FINAL REPORT

Measurement and Modeling of Ecosystem Risk and Recovery for In Situ Treatment of Contaminated Sediments

SERDP Project ER-1552
Phase I

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14. ABSTRACT

The project [ER 1552] addresses strategies to assess the ecological recovery after in-situ sediment treatment by activated carbon (AC) amendment at Hunters Point, San Francisco Bay, California. Rapid assessment tools to measure polychlorinated biphenyls (PCBs) sediment pore water concentrations were tested to correlate aqueous concentrations of PCBs with reduced bioavailability. Polyethylene sampling devices (PEDs) and a PCB immunoassay technique were compared and the polyethylene results correlated with those obtained using conventional methods. Further work with thin polyoxymethylene (POM) sampling devices demonstrated the utility of this method to measure the vertical pore water profile in the sediment and how this compares to the AC distribution in sediment cores. A heat and mass transfer model using field temperature profile data was developed to estimate pore water movement in the intertidal mudflat sediment with AC. Three regional surveys determined the benthic species recruitment pool from 30 reference sites and PCB-induced changes at Hunters Point by means of functional ecology comparisons. A biodynamic modeling approach was developed to predict the uptake of PCBs by benthic invertebrates with different feeding strategies. Field-related influences on bioaccumulation and the effectiveness of AC-amendment were tested with in-situ bioassays and passive sampler deployment. Finally, the biodynamic model was used to simulate PCB exposure scenarios present at Hunters Point and the reference sites. Though PCB tissue concentrations are expected to remain slightly higher at Hunters Point than at the reference sites, the model suggests that the expected remedial success with AC-amendment would comply with the clean-up goal for Hunters Point and sediment quality guidelines.

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(ii) List of acronyms

AC	Activated carbon
AE	Assimilation efficiency
AMW	Artificial marine water
DDT	Dichloro-diphenyl-trichloroethanes
ERL	Effective Range Low
ERM	Effective Range Median
FR	Filtrations rate
GC-μECD	Gas chromatography - micro electron capture detector
HPAH	high molecular weight polyaromatic hydrocarbon
HOC	Hydrophobic organic compound
IR	Ingestion rate
LPAH	low molecular weight polyaromatic hydrocarbon
LOQ	Limit of quantification
MDL	Method detection limit
PAH	Polyaromatic hydrocarbon
PCB	Polychlorinated biphenyl
PE	Polyethylene
PED	Polyethylene devices
PFC	Perfluorinated chemical
POM	Polyoxymethylene
PRC	Performance reference compound
RMSE	Root Mean Squared Error
SPME	Solid Phase Micro Extraction
SERDP	Strategic Environmental Research and Development Program
SOP	Standard operational procedure
SPMD	Semi-permeable membrane device
SQG	Sediment Quality Guidelines
US EPA	United States Environmental Protection Agency

(iii) List of tables

(iv) List of figures

(v) Acknowledgements

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I. Abstract

The project [ER 1552] addresses strategies to assess the ecological recovery after in-situ sediment treatment by activated carbon (AC) amendment at Hunters Point, San Francisco Bay, California. Rapid assessment tools to measure polychlorinated biphenyls (PCBs) sediment pore water concentrations were tested to correlate aqueous concentrations of PCBs with reduced bioavailability. Polyethylene sampling devices (PEDs) and a PCB immunoassay technique were compared and the polyethylene results correlated with those obtained using conventional methods. Further work with thin polyoxymethylene (POM) sampling devices demonstrated the utility of this method to measure the vertical pore water profile in the sediment and how this compares to the AC distribution in sediment cores. A heat and mass transfer model using field temperature profile data was developed to estimate pore water movement in the intertidal mudflat sediment with AC. Three regional surveys determined the benthic species recruitment pool from 30 reference sites and PCB-induced changes at Hunters Point by means of functional ecology comparisons. A biodynamic modeling approach was developed to predict the uptake of PCBs by benthic invertebrates with different feeding strategies. Field-related influences on bioaccumulation and the effectiveness of AC-amendment were tested with in-situ bioassays and passive sampler deployment. Finally, the biodynamic model was used to simulate PCB exposure scenarios present at Hunters Point and the reference sites. Though PCB tissue concentrations are expected to remain slightly higher at Hunters Point than at the reference sites, the model suggests that the expected remedial success with AC-amendment would comply with the clean-up goal for Hunters Point and sediment quality guidelines.

II. Objectives

The project's objectives are to establish correlations between physiochemical measurement tools (i.e., passive sampling devices) and bioavailability in the field to allow confident use of these rapid, inexpensive tools as an additional line of evidence to assess trends in contaminant availability affecting the ecological recovery of a contaminated sediment site after treatment and during a monitored recovery. First, the site-specific correlation between bioavailability and contaminant availability to pore water has to be established with dual deployments of bioassays and passive samplers. Then, sediment managers and DoD users can use passive samplers as a surrogate contaminant availability that allows large-scale risk assessment for the entire site.

Furthermore, a conceptual framework is developed to predict ecosystem recovery, given some knowledge of chemical properties of the sediment, basic information about biodynamics for the contaminants of interest, and data about the benthic community composition of reference sites in the recruitment pool. The conceptual framework integrates biodynamic modeling to assess the required clean-up level that enables recovery. The framework helps to estimate the required reduction in bioavailability (assimilation efficiency) by sediment remediation to reduce bioaccumulation to levels representative of reference conditions. The knowledge of PCB-induced degradation of the benthic community composition is linked to PCB body burden and different exposure scenarios to simulate expected recovery potential at Hunters Point after AC-amendment.

The objectives of the individual milestones are summarized below.

1. Demonstrate the use of rapid assessment tools in the field and correlate with conventional methods

1.1 Characterize (in the laboratory) relevant parameters for the use of polyethylene sampling devices

A series of laboratory tests will be used to measure the rate of uptake of PCBs in polyethylene (PE) sampling devices and the exchange rate with a performance reference compound (PRC). The data will be evaluated by an uptake kinetic partitioning model.

1.2 Apply PEDs and PCB immunoassay in the field at treated and untreated sites

Alternative, rapid tools to measure sediment pore water PCB concentrations in response to in-situ treatment will be introduced and correlated with conventional methods. The measurements using polyethylene devices (PEDs) and an immunoassay procedure will be correlated to PCB pore water concentrations. Alternative methods will be compared to conventional methods in the field and the feasibility of the alternative methods will be assessed.

1.3 Quickly and effectively measure pore water PCB concentration in the field

The suitability of kinetic modeling methods to determine total PCB (63 congeners plus 26 co-eluting groups) concentrations in pore water using data from PEDs exposed to field sediments at Hunters Point Shipyard will be explored. Using PEDs, sampling rates from sediment will be measured in the laboratory and the field to calculate PCB concentrations using two modeling methods. A tested PED kinetic uptake method will be

used to determine the effectiveness of AC amendment to reduce pore water concentrations in sediment at Hunters Point Shipyard.

Furthermore, a partitioning method to measure pore water concentration in a vertical sediment profile will be developed. Pore water measurements with PEDs and polyoxymethylene (POM) devices will be used to monitor the change in pore water concentrations at the sediment surface, and through the AC amended layer and into the untreated sediment.

1.4 Demonstrate that PCB pore water concentrations are indicators of mass transfer to activated carbon particles

Results of the physiochemical tests and biological experiments will be used to assess the reduction in pore water or tissue concentration and to demonstrate that pore water concentration is an indicator of the extent of mass transfer of sediment associated PCBs to AC particles.

2. Conduct regional surveys to determine the benthic species recruitment pool for Hunters Point

2.1 Define recruitment pool of benthic species, along with model and data needs

Benthic sampling surveys in the San Francisco Bay were conducted to determine the benthic community structure that exists in similar physical environments as at Hunters Point but in the absence of contaminant-induced stress. The benthic community structure and the functional make-up of the community were used to identify the natural and seasonal variability of the recruitment pool that would be available for a recovered Hunters Point benthic habitat, and serve as reference conditions to predict recovery at Hunters Point. The hypothesis for this portion of the study was that the benthic community at Hunters Point would not include those species most likely to be stressed by the contaminants due to their feeding mode, reproductive mode, and exposure to and position in the sediment. In other words, the question is, not what species are at the Hunters Point, but rather what species are at the reference sites but not at Hunters Point and their absence can be explained by their functional feeding, reproductive, and exposure/position group. It is hypothesized that the following functional characteristics are least likely to be found in species in the benthic community at Hunters Point:

- Species that must consume sediment or must prey on species that consume sediment.
- Species who expose their reproductive products to the highest contaminant concentrations, e.g., those species that lay their eggs on the sediment surface.
- Species who have little or no barrier between their soft tissue and the sediment.

3. Biodynamic modeling to predict PCB uptake by benthic organism

3.1 Conduct laboratory experiments to define physiological coefficients for biodynamic model

A biodynamic model will be parameterized for species with different feeding strategies to predict PCB tissue concentrations at Hunters Point and to develop a mechanistic understanding of the effects of AC treatment on bioaccumulation. The established model will be used in the following to perform predictions of the ecosystem recovery potential at Hunters Point.

This task has been extended in Phase II of this SERDP project (ER-1552) to investigate adverse biological effects that might be caused by AC addition to sediment.

3.2 Demonstrate applicability of biodynamic modeling in the field

Parallel laboratory and in-situ bioassays will be performed to assess field-related influences on bioaccumulation and to evaluate environmental effects on the performance of an AC-amendment using standard caged organisms testing protocols and passive sampler devices. The biodynamic model will be applied to identify and quantify some of the field-related effects.

4. Development and application of a predictive general ecosystem recovery model

4.1 Predict ecological recovery and compare to field studies

A general concept of ecosystem recovery potential will be developed that combines PCB availability, bioaccumulation, and functional ecology. PCB tissue concentrations for organisms of different functional groups will be estimated with the biodynamic model under different exposure conditions. This ecotoxicological approach will link information of the benthic community structure and PCB exposure to evaluate the expected remedial success of an AC-amendment at Hunters Point.

4.2 Correlate conventional alternative assessment methods and model ecosystem recovery in the field

Analyses using alternative assessment methods like diversity and total abundance, will be compared to the approach of using functional ecology and biodynamic modeling to assess ecosystem change.

5. Monitor and assess carbon amendment in Grasse River

Prior to the September/October 2006 addition of activated carbon to the study site in the Grasse River, pre-treatment monitoring was carried out under the auspices of Alcoa and Dr. Upal Ghosh at the University of Maryland - Baltimore County. Post-treatment monitoring was performed August/September 2007. Researchers of Stanford advised on field methods and reviewed portions of the activated carbon pilot study report.

5.1 Bioaccumulation studies with *Lumbriculus variegates*

Reduction in PCB uptake by the aquatic oligochaete *Lumbriculus variegatus* from Grasse River sediment amended with activated carbon will be demonstrated and the biodynamic model will parameterized to predicted PCB uptake.

5.2 PCB mass transfer modeling with heterogeneous mixing

A model will be investigated to identify and quantify possible advective sediment pore water movement in an intertidal mudflat as at South Basin, Hunters Point. The vertical interstitial pore water flow velocities and mechanical dispersion coefficients will be estimated by heat transport analysis. The quantified pore water movement will be used to determine its relative importance in PCB mass transfer as compared to diffusive mass transfer. This task has been extended in the Phase II of this ER-1552 to further investigate long-term performance of AC-amendment under quiescent field conditions with slow mass transfer compared to well-mixed conditions in the laboratory.

III. Background

1. Demonstrate the use of rapid assessment tools in the field and correlate with conventional methods

1.1 Characterize (in the laboratory) relevant parameters for the use of polyethylene sampling devices

Passive samplers that comprise various polymers have been used to screen for organic chemicals in aquatic environments (Huckins, Petty et al. 2006). Recently, advances in the methodologies to measure and model passive organic chemical uptake by such samplers have resulted in more consistent application and more reliable measurement of environmental aqueous concentrations. The different polymer materials utilized for these passive samplers dictate the individual properties of each sampler and thus differences in uptake rates as well as ease of application in the field. For example, Adams et al. (Adams, Lohmann et al. 2007) concluded that PE is a practical material with faster time to equilibrium and less fouling than with semi-permeable membrane devices, SPMDs, since only a single layer of plastic is exposed on both sides. Cornelissen et al. (Cornelissen, Pettersen et al. 2008) compared five passive samplers and concluded that the selection of a specific passive sampler depends on the objective of the study. If fast equilibrium and low detection limits are required a thin polyoxymethylene, POM (thickness of 55 mm), is advantageous and it has a greater chemical capacity compared to other thin samplers like solid phase micro extraction fibers, SPME fibers. However, if low detection limits are not required and measurements need to be made quickly, either PE, SPME fibers, or thin POM can be used. Further laboratory characterization will facilitate the development of standard operating procedures (SOPs) for the deployment of passive sampler in sediments to assess pore water concentrations.

1.2 Apply PEDs and PCB immunoassay in the field at treated and untreated sites

Conventional ecological assessment methods, such as benthic organism bioassays and community surveys, can provide important information on ecosystem recovery at a contaminated site post-treatment, but the resolution of the information in space and time is limited by the time and cost of sample analysis. Polyethylene devices, PEDs, and a rapid PCB immunoassay have been proposed as complementing tools. They are expected to provide faster, less expensive, but yet reliable measurement.

It is hypothesized that the measurements using PEDs and immunoassays could be correlated to PCB pore water concentrations, which may be linked to the bioavailable fraction of PCBs. In this SERDP project, the measurements using these alternatives will be compared with those using conventional methods in the field and the feasibility of the alternative methods to complement bioassays will be assessed.

1.3 Quickly and effectively measure pore water PCB concentration in the field

In this study, polyethylene devices (PEDs) were chosen since they are inexpensive, commercially available, robust samplers, and easily deployable in sediment. High concentrations of HOCs can be concentrated in PEDs during short field deployments.

Several studies have used PEDs to measure HOC concentrations in water (Booij, Hoedemaker et al. 2003; Huckins, Petty et al. 2006), but no detailed studies have reported on the deployment of PEDs to assess pore water concentrations from passive sampler uptake when our SERDP project started. For this study, models were applied to PED uptake data to predict in a novel way the concentration of polychlorinated biphenyl (PCB) compounds in pore waters using field PED measurements. Such pore water concentrations, in comparison to total sediment concentrations, are a more appropriate indicator of bioavailable contamination in an ecosystem (Cornelissen, Breedveld et al. 2006; Ehlers and Loibner 2006; Sun and Ghosh 2008). Currently, HOC concentrations in pore waters in the field are difficult to measure because large volumes of water are needed for HOC detection, which is confounded by large amounts of colloids in pore water (Ter Laak, Agbo et al. 2006). Therefore, fast and predictive measures of contaminant availability in sediments are greatly needed for risk assessment and remedial treatment evaluation.

Comparison with POM and vertical pore water profiles

Work on comparison of thin polyethylene and thin polyoxymethylene sediment pore water samplers is continued through collaboration with Stanford University and Amy Oen from the Norwegian Geotechnical Institute. The continuation of this work at the Hunters Point field site provides a special opportunity to follow the long-term ecosystem risk and recovery of a contaminated site after in-situ treatment through in-situ measurement of PCBs in sediment profiles. These studies will build on the work of Tomaszewski et al. (2008) by utilizing passive samplers as rapid measurement tools to measure concentrations of PCBs in the pore water phase. Freely dissolved concentrations in pore water are a useful parameter for risk assessment, as these measures are directly related to amounts of chemicals in organisms and are therefore a much better indicator for risk than measures of total concentrations in sediment.

Vertical pore water profiles measured in the field with passive samplers will be used to monitor the change in concentrations at the sediment surface, and through the AC-amended layer and down into the untreated sediment. Sediment cores were taken and thin slices (~2cm) were analyzed for AC content.

1.4 Demonstrate that PCB pore water concentrations are indicators of mass transfer to activated carbon particles

Activated carbon acts as a strong sorbent for hydrophobic organic compounds like PCBs. The sequestration of PCBs reduces their availability to biota and water. Hence, a reduction in pore water or tissue concentration may be used as an indicator of the extent of mass transfer of sediment associated PCBs to AC particles.

David Werner at the University at Newcastle conducted a comprehensive review of relationships among bulk sediment, aqueous, and tissue (lipid) concentrations for a range of native black carbon concentrations. The native black carbon is a strong sorbent in the natural sediment, and review of sorption data for sediment with black carbon allows us to study sequestration and response of pore water similarly to activated carbon. The SERDP project ER-1552 supported part of the work for Werner's and the findings are reported in Werner et al. (2009, Appendix A). Briefly, traditional and new relationships of PCB distribution among the solid phases, the free aqueous phase, and

biolipids were comprehensively reviewed using seven well characterized freshwater and marine sediments polluted with PCBs. The traditional relationship relating free aqueous concentration and biolipid concentration to sediment total organic carbon, compound octanol-water partitioning coefficient, and solid-phase contaminant concentration overestimates measured free aqueous concentrations and biolipid concentrations by mean factors of 8 and 33, respectively. By contrast, relationships based on measured free aqueous phase concentrations or the PCB mass fraction desorbed from sediment provides reasonable predictions of biolipid concentrations. Solid phase concentration-based predictions perform better when sorption to amorphous organic matter and black carbon (BC) is distinguished. Contrary to previously published relationships, BC sorption appears to be linear for free aqueous PCB-congener concentrations in the picogram to microgram per liter range.

2. Conduct regional surveys to determine the benthic species recruitment pool for Hunters Point

2.1 Define recruitment pool of benthic species, along with model and data needs

Contaminants in sediment that cause chronic toxicity to individual organisms can simplify the community structure by reducing the abundance of sensitive species and increasing the abundance of tolerant species (Gray 1979; Pearson, Gray et al. 1983; Hyland, Balthis et al. 2003; Fuchsman, Barber et al. 2006). Even though, general theories are being developed to use information of the benthic community composition to evaluate ecosystem integrity, a mechanistic understanding of the physiological and environmental characteristics involved is still vague (Ugland and Gray 1982; Pachepsky, Crawford et al. 2001). Despite the evidence of ecological risk, there are few ecologically-based decision-making tools available to assess the effectiveness of remediation of sediment contamination. While the response of one species in an ecosystem may not necessarily be representative of the response of the whole system, changes in the species groups performing a common ecosystem function may be represent a response to contamination. For example, shifts in benthic composition or abundance of functional feeding groups have been observed along contaminant gradients in some systems (Pearson and Rosenberg 1978; Horne, Finley et al. 1999; Thompson and Lowe 2004). For example, functional feeding groups allow us to group species by the different modes of acquiring food. These feeding modes determine an animal's exposure to chemical pollution due to differing physical exposure to the contaminated environment. A benthic survey study should target the identification of species which are at reference sites but not the contaminated site. This approach can then evaluate the absence of those species by their functional feeding, reproductive, and exposure/position groups. Relating the abundance of benthos by functional groups to sediment contamination presents an ecotoxicological approach, which integrates aspects of ecology and toxicology to evaluate the status of an ecosystem (Chapman 2002).

A careful choice of reference sites and a detailed understanding of a non-impacted (ambient) benthic community structure are crucial for the study design. The physical habitats of the reference sites have to match the site of interest (e.g., salinity, grain size distribution, TOC content, tidal regime, water depth, temperature etc.). The reference

sites must not be impacted by pollution that is above background levels of the ecosystem. Seasonal and natural variability of the benthic community has to be assessed by repeated surveys. The probability of species recruitment in a 'healthy' environment is primarily controlled by salinity, physical habitat, and availability of larvae. Thus these factors must be considered in the design for an ecosystem recovery study. For example, larvae may be transported from a widely distributed recruitment pool to the study area and thus broad areas should be examined. Re-colonization of the contaminant-sensitive species within functional groups in a recovering habitat reflects the success of the in-situ remediation approach.

3. Biodynamic modeling to predict PCB uptake by benthic organisms

Biodynamic models describe the uptake of contaminants as a mass balance of uptake from water, uptake from sediment, loss rates, and growth rates. Luoma and Rainbow (2005) recently proposed biodynamics as a unifying concept in metal bioaccumulation. The biodynamic modeling technique ultimately provides the basis for a rational procedure to predict organism uptake, to assess and predict bioavailability of PCBs to benthic organisms, and to evaluate risk to the benthic community. Hence, this modeling framework was used to compare PCB bioaccumulation of species with different feeding strategies and to compare different levels of exposure to evaluate the recovery potential following in-situ treatment that reduces PCB bioavailability.

3.1 Conduct laboratory experiments to define physiological coefficients for biodynamic model

The biodynamic model was previously established for metal bioaccumulation (Griscom, Fisher et al. 2002). The model has been modified to describe uptake of organics and the affect of AC on reducing exposure by ingestion and dermal processes:

$$\frac{d\,C_{org}}{d\,t} = C_{sed} \cdot IR \cdot \left[(1-f)\cdot AE_{sed} + f \cdot AE_{AC}\right] + C_w \cdot k_w - C_{org,\,t} \cdot (k_e + k_g) \quad \textbf{(1)}$$

With C_{org} the PCB conc. in the organism (µg/g dry tissue); C_{sed} the PCB conc. in the sediment (µg/g dry weight); IR the ingestion rate (g particles/g dry weight / d); AE_{sed} and AE_{AC} the PCB adsorption efficiency from the sediment and activated carbon, respectively; f the fraction of PCB associated with sediment; k_w the aqueous uptake rate constant (L/g per d); C_w the aqueous PCB concentration (µg/L); k_e the rate constant of loss (1/d); and k_g and growth rate constant.

In order to measure and model the PCB-uptake behavior two test species with different feeding strategies have been selected that are native in the San Francisco Bay and representative for the field site at Hunters Point. Clams represent a species that can deposit- as well as filter-feed and the aqueous uptake rate can be described as the product of AE from water and the filtration rate (FR) (L/g per d) (McLeod, Van den Heuvel-Greve et al. 2007). Because the organism has the option to filter-feed, it can regulate its exposure to contaminated sediment.

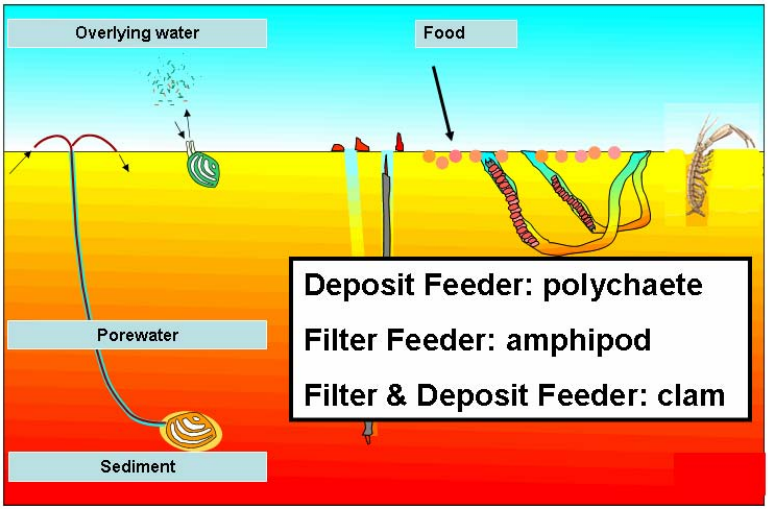

Figure 1. Test organisms with different habitat and feeding strategies within the sediment and overlying water.

The second species, a polychaete, is a representation for a deposit feeder. Deposit-feeding organisms are promising test organisms for sediment quality because they ingest high amounts of the contaminated sediment, i.e., multiple times their body weight per day. The polychaete chosen, *Neanthes arenaceodentata,* is sessile and representative for the local pollution at Hunters Point. Millward et al. (2005) showed that the PCB bioaccumulation in this polychaete, *Neanthes arenaceodentata,* is reduced by 82% when exposed for 28 days to Hunters Point sediment 1-month after AC amendment.

In order to apply the biodynamic model for this polychaete, time-series bioassays and individual exposure experiments have to be conducted to parameterize the model for a better mechanistic understanding of bioaccumulation and the effects of AC for these species.

3.2 Demonstrate applicability of biodynamic model in the field

The goal of sediment remediation, whether by removal, capping, or in-place treatment, is to mitigate risk by reducing exposures to aquatic biota and humans. The goals of this task were to identify differences in PCB exposure from sediment under field and laboratory conditions using caged macroinvertebrates and to evaluate the effectiveness of an in-place sorbent treatment under similar test conditions.

Previous sediment bioassays have shown that AC-amendments can significantly reduce bioaccumulation under controlled laboratory conditions (Millward, Bridges et al. 2005; McLeod, Luoma et al. 2008; Sun, Werner et al. 2009; Janssen, Croteau et al. 2010). In laboratory studies, biodynamic modeling has been used to predict PCB bioaccumulation and the effects of reduced availability after AC-amendment and quantify uptake from sediment and water (McLeod, Luoma et al. 2008; Sun, Werner et al. 2009; Janssen, Croteau et al. 2010). While laboratory studies are necessary to conduct controlled bioassays and for a mechanistic understanding of bioaccumulation, there is limited simultaneous comparison of ex-situ and in-situ bioassays. Comparative in-situ and ex-situ toxicity studies find that field-related phenomena influence assessments of contaminant exposure and organism survival, growth, feeding and development (Sasson-Brickson and Burton Jr. 1991; Burton Jr. 1995; Hatch and Burton 1999; Besten den,

Naber et al. 2003; Burton, Greenberg et al. 2005; Cho, Ghosh et al. 2009). Likewise, simultaneous laboratory and field bioassays can help define how environmental conditions influence measures of the effectiveness of remediation alternatives. In this regard, biodynamic modeling is useful to explain differences between ex-situ and in-situ experiments by mass balance of influx and efflux to evaluate environmental influences.

Parallel laboratory and in-situ bioassays should be performed to assess field-related influences on bioaccumulation and to evaluate environmental effects on a sediment amendment with AC using standard caged organisms testing protocols and passive samplers. Furthermore, the biodynamic model can be applied to support conclusions of field observations.

4. Development and application of a predictive general ecosystem recovery model

The underlying hypothesis for ecosystem recovery is based on principles of functional ecology. The benthic community at Hunters Point will reflect feeding strategy, position in or on the sediment, reproductive strategy, and physiology. Thus, re-colonizing animals and recovery will be oriented towards community structures of the recruitment pool and reference sites where sediment toxicants are present at ambient levels.

4.1 Predict ecological recovery and compare to field studies

The goal of sediment remediation, whether by removal, capping, or in-place treatment, is to mitigate risk by reducing exposures to biota and humans. Chemical pollution can alter the benthic community composition and remediation should allow for recovery of the local ecosystem functions. To recover benthic community functions, sediment remediation has to consider site-specific biological measures to assess the risk and required remedial success. Measures of exposure like contaminant concentrations in the bulk sediment, pore water, and tissue of collected or transplanted organisms are typically used in risk assessment before remediation and during monitoring. However, the evaluation of remediation success is challenging because little research has focused on the linkage of specific contaminants to responses on the population level, and ecosystem recovery as a response to reduced exposure are still not well understood (Thompson, Adelsbach et al. 2007).

The sediment quality guidelines (SQGs) suggest thresholds of bulk sediment concentration of chemical pollutants above which adverse effects in benthic infaune are more likely to occur (Long and MacDonald 1998). These guidelines are based on various studies that analyzed the biological responses and benthic community changes relative to contaminant concentrations. The SGQs are widely used screening values to identify pollutants of concern but these guidelines do not distinguish between different sediment properties and thus are not site-specific. Reference conditions at sites with ambient sediment contamination in the same watershed should be considered to define remedial goals.

Risk assessment at Hunters Point, South Basin, focuses on pollution mainly by polychlorinated biphenyls (PCBs) and some heavy metals (Battelle 2004). PCBs are persistent, hydrophobic, and bioaccumulative contaminants, which are widely distributed, and listed as probable carcinogens to humans, and account for the second-leading cause

of fish consumption advisories in the U. S. (EPA 1996; EPA 1997). While aqueous PCB concentrations in overlying water even at polluted sites are in general below the acute toxic level (Fuchsman, Barber et al. 2006), chronic effects on growth, reproduction, survival, life span, or teratogenic effects in wildlife, can be observed at lower concentrations (Bridgham 1988; Landrum , Faust et al. 1989; Borgmann, Norwood et al. 1990; Murdoch, Chapman et al. 1997; Fisher, Chordas et al. 1999; Zeng, Bay et al. 2003). Thus exposure to PCBs associated with sediment poses an ecological risk to benthic invertebrates, fish and birds that forage at the mudflat at Hunters Point.

Adverse effects are related to the degree of exposure, which is defined by the species' interactions with the contaminated environment. It is well recognized in the benthic ecology literature (although less so in the ecotoxicology literature) that measures of community structure such as diversity indices range from insensitive to un-predictive in terms of facilitating understanding of effects of contaminants like PCBs. The bridge between ecotoxicology and ecology has long been recognized as weak in this regard. The underlying hypothesis for our study for ecosystem recovery is based on principles of functional ecology. The benthic community at Hunters Point will reflect an organism's coping strategies regarding their feeding, position (protective strategies such as physical barriers) and reproduction mode. Recovery will be oriented towards community structures of the recruitment pool and references sites where sediment toxicants are present at ambient levels. The benthic ecological recovery concept uses the species-specific physiology of PCB exposure to facilitate our understanding of ecological recovery potential after sediment remediation.

The demonstration of reduced tissue concentrations in benthic organisms after remediation at Hunters Point has to be further linked to reference conditions to evaluate the expected ecological benefit for the benthic community structure and to answer the questions about "how much treatment is (safe) enough?". Most parts of the San Francisco Bay experience chemical impacts, which are reflected by changes in the benthic composition (Battelle 2004; Thompson and Lowe 2004; Davis, Hetzel et al. 2007). Appropriate reference conditions reflect the biological potential of the ecosystem considering natural variability (Thompson and Lowe 2004; Carter, Purcell et al. 2009).

4.2 Correlate conventional alternative assessment methods and model ecosystem recovery in the field

Contaminants in sediment that cause chronic toxicity to individual organisms can simplify the community structure by reducing the abundance of sensitive species and increasing the abundance of tolerant species (Gray 1979; Pearson, Gray et al. 1983; Hyland, Balthis et al. 2003; Fuchsman, Barber et al. 2006). Even though general theories are being developed to use information about the benthic community composition to evaluate ecosystem integrity, a mechanistic understanding of the physiological and environmental characteristics involved is still vague (Ugland and Gray 1982; Pachepsky, Crawford et al. 2001). Despite the evidence of ecological risk, there are few ecologically-based decision-making tools available to assess the effectiveness of remediation of sediment contamination.

Diversity indices and the abundance of indicator species are regularly applied to evaluate effects of pollution on the benthic community (Dolah, Holland et al. 1999; Hunt, Anderson et al. 2001; Llanso, Volstad et al. 2009). These measures sometimes serve as

useful tools to identify changes of the community composition, but are often not sensitive enough to recognize degradation of ecosystem processes (Dauer, Luckenbach et al. 1993; Horne, Finley et al. 1999; Hyland, Balthis et al. 2003). In addition, indices can be problematic in systems that are naturally physically stressful, which limits the number of species that are capable of thriving in the environment.

Adverse effects in wildlife are related to exposure to pollutants, which is a function of the species' interactions with the contaminated environment. Basing ecosystem recovery on principles of functional ecology offers a promising assessment method to identify pollution-induced changes in the benthic community structure (Baird, Rubach et al. 2008; Burton and Johnston 2010). Functional-based analysis offers a more complete description of the benthic community by grouping species by their functional traits in addition to taxonomic analysis. This is also referred to as trait-based ecological risk assessment (TERA) (Baird, Rubach et al. 2008). While the response of one species in an ecosystem may not necessarily be representative of the response of the whole system, changes in the species groups performing a common ecosystem function represent a serious alteration in the ecological processes. Analysis by feeding traits is especially interesting because feeding groups reflect differences in dietary routes and hence exposure to the contaminated environment (Pearson and Rosenberg 1978; Horne, Finley et al. 1999; Thompson and Lowe 2004). In such circumstances, biodynamic modeling can estimate bioaccumulation and the mechanistically reconcile dietary routes with contaminant exposure (McLeod, Luoma et al. 2008; Sun, Werner et al. 2009; Janssen, Croteau et al. 2010).

5. Monitor and assess carbon amendment in Grasse River

Prior to the September/October 2006 addition of activated carbon to the study site in the Grasse River, pre-treatment monitoring was carried out under the auspices of Alcoa and Dr. Upal Ghosh at the University of Maryland - Baltimore County. Post-treatment monitoring was performed August/September 2007. Stanford University researchers developed and evaluated a biodynamic model for describing the uptake of PCBs by a fresh water clam and validated the model with Grasse River sediments. Laboratory data were collected for extending this modeling approach to a freshwater oligochaete.

5.1 Bioaccumulation studies with *Lumbriculus variegates*

Work involving the Grasse River pilot scale study site (Massena, NY) complements the remainder of work under ER-1552 focusing on Hunters Point. Research completed under Task 5.1 extends our ability to predict PCB uptake by aquatic organisms through understanding how activated carbon amendment impacts bio-uptake by the aquatic oligochaete *Lumbriculus variegatus*. *L. variegatus* is the only freshwater benthic invertebrate that has been selected as a bioaccumulation test organism by the USEPA [USEPA 2000]. Prior work in the Luthy laboratory characterized the biodynamics of PCB uptake (1) from Hunters Point sediment by the estuarine clam *Macoma balthica*, and (2) this was extended with SERDP support to describe the biodynamics of PCB uptake from Grasse River sediment by the freshwater clam *Corbicula fluminea* [McLeod, 2008].

5.2 PCB mass transfer modeling with heterogeneous mixing

Sediments accumulate hydrophobic organic compounds (HOCs) such as polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and dichloro-diphenyl-trichloroethane (DDT). Sediments thus act as reservoirs, exposing HOCs to benthic biota, releasing HOCs into pore water, and contributing HOCs to the aquatic food web. Recent research shows that certain sediment particle types, known as black carbon, may have stronger sorption capacity than inorganic particles with coatings or inclusions of natural organic matter (Ghosh, Zimmerman et al. 2003). Char, charcoal, soot and their derivatives are such types with strong sorption capacity. Once the HOCs are sorbed into the BCs, they become much less available than HOCs sorbed on other mineral-based particles (Ghosh, Gillette et al. 2000; Ghosh, Zimmerman et al. 2003).

These findings motivated studies of a novel *in-situ* sediment treatment strategy using activated carbon (AC) as a strong sorbent to sequester HOCs. By incorporating AC into HOC-contaminated sediment, HOCs would be redistributed, sorbed on to AC particles and become less available to pore water and biota. The proof of concept was demonstrated in a series of laboratory and field studies (McLeod, Van Den Heuvel-Greve et al. 2004; Zimmerman, Ghosh et al. 2004; Millward, Bridges et al. 2005; Cho, Smithenry et al. 2007; McLeod, Van den Heuvel-Greve et al. 2007; McLeod, Luoma et al. 2008; Tomaszewski and Luthy 2008; Cho, Ghosh et al. 2009). For instance, introducing 3.4 dry wt % of AC into well-mixed sediment-water slurries in the laboratory yielded about 90% reductions of PCBs, PAHs, and DDTs in water phases and benthic organisms (McLeod, Van Den Heuvel-Greve et al. 2004; Zimmerman, Ghosh et al. 2004; Millward, Bridges et al. 2005; Zimmerman, Werner et al. 2005; McLeod, Van den Heuvel-Greve et al. 2007; McLeod, Luoma et al. 2008; Tomaszewski and Luthy 2008). Mixing about 2% AC into the top 30-cm sediment layer at a mudflat in San Francisco Bay, CA yielded 50-66% reduction in HOC concentrations in pore water, passive samplers and benthic test organisms (Cho, Smithenry et al. 2007; Cho, Ghosh et al. 2009). Other than AC dosage, the difference in performance between laboratory and field trials may have been a result of HOC mass transfer in the field occurring without continuous and complete mixing of AC and sediment. In this case, HOC diffusion between sediment and AC particles may be a limiting process for HOC mass transfer.

To explain HOC mass transfer in a stagnant system, Werner et al. (2006) built a mass transfer model of an unmixed sediment system that considered intra- and inter-particle movement of HOCs by sorption-retarded molecular diffusion. The model predictions showed that the HOC mass transfer to AC in a quiescent system would be greatly retarded and the full effect of reduction in aqueous HOC concentration by AC would be delayed for a number of years (Werner, Ghosh et al. 2006). This model did not account for possible advective pore water movement in an intertidal mudflat (Putnam 1949; Webb and Theodor 1969; Huettel, Ziebis et al. 1998 ; Rocha 2000), which could affect HOC mass transfer. Several mechanisms are believed to result in advective movement in intertidal or subtidal areas: bottom currents, propagating waves, subtidal pumping, bottom water density changes, and bottom microtopography (Rocha 2000; Precht and Huettel 2004). The quantification of the pore water movement is challenging because the intertidal system undergoes rapid changes (Carpenter 1997; Rocha 2000).

To assess advective pore water movement within the upper 0-60 cm sediment layer in a intertidal mudflat, "heat as a tracer" was used (Anderson 2005). The use of heat as a tracer of vertical groundwater flow has been a standard tool in hydrogeology (Anderson 2005), and remains a powerful tool in cases with predominantly vertical fluid and heat transport (Anderson 2005; Constantz 2008). Such a case was the nearly horizontal intertidal mudflat of this study (slope approx. 0.25-0.5%, estimated from Nautical Chart [NOAA]). The low local relief and lack of significant inland water drainage imply very low horizontal hydrologic gradients at this site. The sediment surface temperature of intertidal areas fluctuates because of tidal action and solar radiation (Cho, Kim et al. 2005; Moffett, Tyler et al. 2008). Heat transport by conduction and convection determines the temperature depth profiles in the sediment.

The pore water movement determined by inverting a heat transport model was used to assess its relative importance in HOC mass transfer compared to diffusive mass transfer. An innovative aspect of this study was its combination of: (a) the inverse approach of (Silliman, Ramirez et al. 1995) to quantify the vertical pore water flow beneath a water body of a known temperature with (b) the effects of ephemeral surface flooding on vertical fluid and heat flow (Constantz and Thomas 1997) and (c) sediment temperature profile data from the challenging environment of an intertidal mudflat. This approach differs from previous descriptive studies of sediment temperature profiles in mudflats (Harrison and Phizacklea 1987; Cho, Kim et al. 2005) and studies that specified *a priori* a specific mechanism of pore water convection prior to calculating advective velocities (e.g., thermal density instability (Rocha 2000). Another innovative aspect of this study was its use of heat as a tracer for understanding HOC transport in the environment.

The presented work used field-collected temperature data and a heat transport model to estimate limiting values of vertical interstitial pore water flow velocities and mechanical dispersion coefficients. Then this information was applied to calculate a ratio of the rate of advection to the rate of molecular diffusion (a dimensionless Peclet number) as well as a ratio of the mechanical dispersion to the molecular diffusion. The study concludes with a discussion of the relative influences of advection, mechanical dispersion, and molecular diffusion processes on PCB mass transfer.

IV. Material and Methods

Throughout this report, PCB concentrations are reported as total PCBs based on a modified EPA method as described by Ghosh et al. (Ghosh, Zimmerman et al. 2003), where 92 individual congeners and co-eluting congeners are measured (unless otherwise mentioned).

1. Demonstrate the use of rapid assessment tools in the field and correlate with conventional methods

1.1 Characterize (in the laboratory) relevant parameters for the use of polyethylene sampling devices

PED Preparation and Extraction. Low-density polyethylene (PE) with no additives and a thickness of 51 μm was obtained from Brentwood Plastics (St. Louis, MO). The PE was pre-cleaned with a series of solvents (hexane, methanol, deionized water) then allowed to dry at 60 ^{o}C for 4 hours. PCB concentrations in the PEDs used in this study were measured by cutting the deployed PE from frames, rinsing with deionzed water, wiping dry with a Kimwipe, and extracting in 40 mL hexane. Surrogate standards 3,5-dichlorobyphenyl (PCB 14) and 2,3,5,6-tetrachlorobiphenyl (PCB 65) were spiked into the hexane at the beginning of extraction. After 24 hours of extraction with rotation at 2 rpm, the PE was removed from hexane, rinsed with solvent, and allowed to dry for weight determination.

Field Time Series. PEDs were constructed by vertically attaching three 2.5 cm wide pre-cleaned PE strips to a 10 by 30 cm rectangular frame made of stainless steel tubing. In January 2007, a total of 5 PEDs were placed in sediment at a depth of 5-15 cm near the middle of Plot C or Plot D, in a 30 cm radial pattern. The deployed PEDs were retrieved over time (15, 30, 58, 155, and 239 days) to assess the approach towards equilibrium.

Laboratory Time Series. Laboratory calibrations were used to determine elimination rates of performance reference compounds (PRCs) and uptake rates of PCBs in PEDs over time under well-mixed conditions. Wet sediment (400 g) collected in January 2007 from Plots C or D was placed in 1 L glass bottles, and the bottles were filled with 30 ‰ artificial saltwater. The sediment slurries were rotated at 2 rpm at 15 ^{o}C for two weeks, then three PRC-spike PE strips (Plot C: 2.5 by 5 cm, Plot D: 2.5 by 10 cm) were added in each bottle. At intervals (0.4, 1, 2, 7, 14, 28, 60, and 99 days), bottles were removed from mixing, and the PE strips were removed and extracted.

Modeling. Work by others indicates uptake rates of HOCs in SPMDs and PEDs are governed by the aqueous boundary layer under low flow conditions for solutes in the log K_{ow} range from 4.4-8.0 (*3, 17–19*). Booij et al. (Booij, Hoedemaker et al. 2003) provide an additional modeling approach for stagnant sediments that incorporates aqueous diffusion coefficients and sediment characteristics. Reviews of passive sampling theory can be found elsewhere (Huckins, Petty et al. 2002; Booij, Hoedemaker et al.

2003; Huckins, Petty et al. 2006). The kinetic models used in this study assume the rate of desorption from the sediment is not rate limiting and uptake kinetics are governed by the aqueous boundary layer. The diffusive mass transfer of an HOC in a PED, regardless of equilibrium or kinetic sampling, can be expressed as

$$C_{PED} = K_{PEW} C_W [1-\exp(-k_e{}^t)] \quad \textbf{(2)}$$

where C_{PED} (ng/kg) is the amount of HOC in a PED, K_{PEW} (L/kg) is the PED-water partition coefficient, C_w (ng/L) is the aqueous concentration, and k_e (1/d) is the exchange rate coefficient (Booij, Hoedemaker et al. 2003). In this study, *K*PEW was estimated from an expression in Booij et al. (Booij, Hoedemaker et al. 2003) for low-density polyethylene data measured at similar salinity and temperature conditions, and values are presented elsewhere (Supporting Information (Tomaszewski and Luthy 2008)). The sampling rate (R_S, L/d) is the apparent water volume extracted per unit time by the sampler and is given by

$$k_e = \frac{R_S}{K_{PEW} M_{PE}} \quad \textbf{(3)}$$

where M_{PE} (kg) is the mass of PED. The exchange rate coefficient (k_e) for congener uptake can be estimated by conducting a time series test to measure uptake rates. The increase of HOC concentrations can be measured over time, and the results fitted to a form of equation 2:

$$k_e = (-1/t)\ln\left[1 - \frac{C_{PED}(t)}{K_{PEW} C_w}\right] \quad \textbf{(4)}$$

where C_w is the equilibrium water concentration and $C_{PED}(t)$ denotes congener concentrations in PEDs over time. The effect of environmental variables (hydrodynamics and biofouling) on the uptake rates of HOCs, as controlled by the aqueous boundary layer, are closely approximated by the effect on the loss rates of impregnating performance reference compounds (PRCs), which are analytically non-interfering, model compounds of low to moderate K_{OW} (Huckins, Petty et al. 2002). As the amount of PRCs in PEDs before and after deployment can be measured, the rate of elimination can be calculated by

$$k_{e,PRC} = \ln\left(\frac{C_{PED,PRC}(0)}{C_{PED,PRC}(t)}\right)\left(\frac{1}{t}\right) \quad \textbf{(5)}$$

where $C_{PED,PRC}$ (0) is the PRC concentration before deployment (t =0). From the knowledge of $k_{e,PRC}$, methods then can be used to adjust the measured PRC loss parameters ($k_{e,PRC}$ and $R_{S,PRC}$) and estimate sampling rates for HOCs that have K_{OW} values larger than that of the PRC and that observe boundary layer controlled uptake. The use of a few PRCs to estimate sampling rates for numerous HOCs is advantageous in terms of ease, time, and cost for large-scale application of PEDs as assessment tools in the field

Further details on modeling with use of the *molar volume adjustment* and the *exposure adjustment factor* can be found in Tomaszewski and Luthy (Appendix A).

1.2 Apply PEDs and PCB immunoassay in the field at treated and untreated sites

Immunoassay kit. A commercial immunoassay kit was purchased from Strategic Diagnositics Inc. (Newark, DE) (Figure 2). This kit contains polystyrene test tubes, enzyme conjugate solution, antibody coupled paramagnetic particle solution, diluent/zero standard, color solution containing hydrogen peroxide and chromogen, stop solution, washing solution (preserved deionized water), three concentrations of PCB standards, and a control sample. Analysis was performed by following the analytical procedure described in the manual provided by the manufacturer.

Figure 2. Immunoassay Kit from Strategic Diagnostics Inc. (Newark, DE) with example test tubes.

Sediment samples. The Hunters Point sediment samples were collected at various time points from plots C, D, E, and F at the field site. The sediment cores with approximately 1 foot deep were taken from each plot and brought into the laboratory. Two sample sets, one taken in 2007 and the other in 2008, were prepared as composite. Another sample set collected in 2007 was prepared by dividing cores into the upper and the bottom portions, six inches respectively. All the sediment samples were completely air-dried and ground for the further analysis.

Sample preparation. Water samples can be directly analyzed using the immunoassay kit with proper dilution of the sample. However, the sediment sample should be extracted prior to the immunoassay analysis. The sample extraction kit was purchased from Strategic Diagnostics Inc. for practice purposes only. After the training, the sediment samples were processed in the laboratory using laboratory supplies. The detail of the sediment sample extractions is given below. 20 ml of methanol was added into 20 ml size vial containing one gram of Hunters Point sediment sample (dried). The

sample was extracted by shaking for one full minute and the sample was allowed to settle for 15 minutes. Clear overlying methanol was collected and filtered using Whatman autovial syringeless filters with PTFE membrane having pore size of 0.45 μm (Fisher scientific, IL). Filtered extract was stored in a vial and 25 μl of the extract was taken and diluted by adding it into 1 ml of methanol. The diluted extract was analyzed as a sample for the immunoassay.

Sample analysis by GC-μECD. The same methanol extracts from the immunoassay analysis was further subjected to GC analysis to correlate PCB concentrations measured by the immunoassay (reported as Aroclor 1254) with PCB concentrations reported from other conventional methods that use GC. Prior to GC analysis, the extract was cleaned up using 3% deactivated silica gel column by following EPA method 3630C.

1.3 Quickly and effectively measure pore water PCB concentration in the field

Field deployment (28 days) of PEDs. For each 28-day study, five PEDs were deployed per plot at randomly assigned locations, as denoted by sampling stations 1-5 in the four treatment plots. Before amendment with activated carbon and 6 months after amendment, PEDs were constructed by cutting pre-cleaned PE into 14.5 cm^2 circles and attached to circular frames made of aluminum or coated wire. The PEDs were placed horizontally in the sediment at depth of 15 cm. For the 18 months after amendment deployment pre-cleaned PE strips were impregnated with 66 ng PCB 29/g PED and 55 ng PCB 69/g PED, as determined by field blanks. PEDs were constructed by horizontally attaching one PRC-spiked PE strip (3.8 cm wide) to a stainless steel frame (10 cm by 30 cm). The PEDs were placed at a depth of 5-15 cm. Upon retrieval, the PE strips were cut in half before extraction, creating a total of ten replicates per plot.

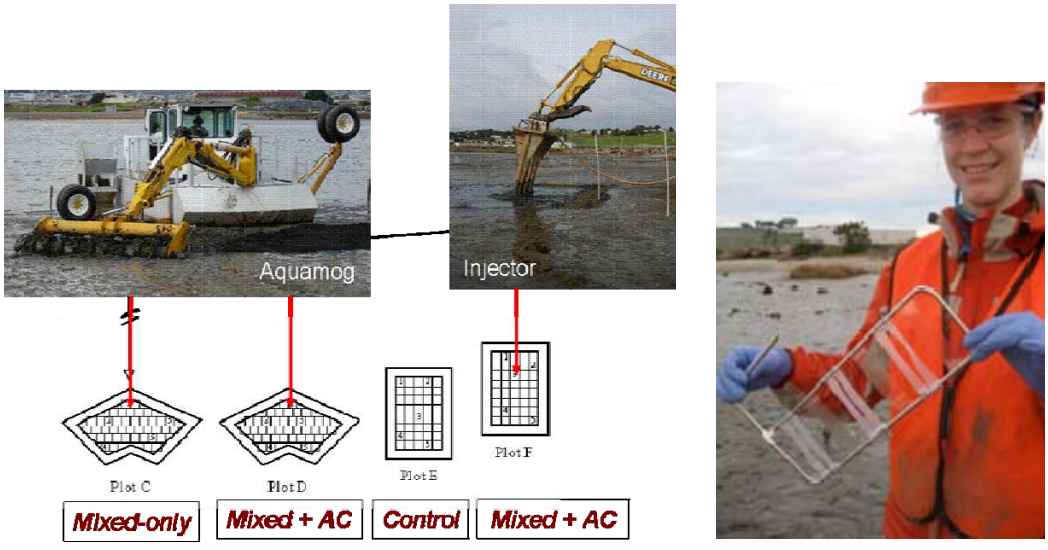

Figure 3. Schematic of test plots and mixing devices (left) and PE devises as deployed in the field (right).

Field-collected Pore Water. In September 2007, pore water from two locations in Plot C was collected at low tide by creating a 10 cm depression in the sediment surface and transferring the pooled pore water to a clean glass jar. The jars were transported to the laboratory in chilled coolers, and the water was centrifuged to remove large particles and transferred to clean vessels using glass pipets. The pore water was then flocculated to remove colloids and extracted as described elsewhere (Ghosh, Weber et al. 2000).

Vertical pore water profile

In addition to deploying polyethylene samplers (PE), ultra thin (17 μm) passive samplers made of polyoxymethylene (POM) were deployed, which have been utilized successfully in Norwegian fjord systems to measure concentrations of PCBs in the overlying water (Cornelissen, Arp et al. 2008). These ultra-thin POM samplers were able to obtain equilibrium in the overlying water within 2 weeks. However, since these samplers are employed to measure pore water concentrations, the time to reach equilibrium was expected to take longer. To monitor the progress towards equilibrium, performance reference compounds (PRCs) were utilized. Tomaszewski et al. (2008) used PCB29 and PCB69 as PRCs. In addition to these, a penta-, a hexa- and a hepta-chlorobiphenyl were included to assess a wider range of PRCs and their dissolution from the passive samplers. A PED-partitioning method is being developed to measure pore water concentration at various sediment depths. These pore water measurements at depth were to monitor the change in pore water concentrations at the sediment surface, through the AC amended layer and down into the untreated sediment.

Figure 4. Pictures (top to bottom): deployment of passive sampler rods for vertical pore water profiles; sediment core; close-up of deployment rod with POM passive sampler.

POM passive samplers were deployed at Hunters Point, Parcel F, at five positions in our treatment plot where 2 to 3 % activated carbon was amended with the Aquamog in January 2006. New deployment rods allow us to measure a vertical profile of the pore water over 40 cm depth in approximately 5 cm intervals. In addition to the deployment, sediment cores were taken at the five deployment positions. The cores were sectioned into one inch slices and are currently being analyzed for total organic carbon. This carbon data provides a reference of the depth profile of carbon at our sampling locations, which was compared to the pore water measurements. The

retrievals of the sample rods were performed after 14, 26, 57, 100, and 154 days of deployment. Five PCBs from different homolog groups, which are not present in the sediment, were used as performance reference compounds (PRCs) to estimate equilibrium between the pore water and the sampler. An increased uptake of PCBs in the passive samplers was observed over the exposure time, as would be expected with slow mass transfer to the samplers.

The pore water concentration can be determined based on the concentration in the passive sampler and knowledge of the passive sampler and compound-specific equilibrium partitioning coefficient, K. At equilibrium

$$C_{ps} = K_{ps} C_{pw} \quad \textbf{(6)}$$

where C_{ps} (ng/kg) is the concentration in the passive sampler, K_{ps} (L/kg) is the passive sampler-water partition coefficient and C_{pw} (ng/l) is the pore water concentration. For the vertical pore water profile measurements with POM equilibrium has been achieved after about 100 days of exposure in the field.

1.4 Demonstrate that PCB pore water concentrations are indicators of mass transfer to activated carbon particles

It was demonstrated that pore water concentrations are indicative of PCB mass transfer to carbon with a combination of experiments including two types of passive samplers tested in the laboratory and deployed in-situ, as well as aqueous equilibrium pore water measurements, and various bioassays with two clams, an oligochaete, and a polychaete. The experiments were designed to assess the change in either pore water concentration or tissue concentrations after AC amendment. A reduction in pore water or tissue concentration is used as an indicator of the extent of mass transfer of sediment associated PCBs to AC particles. A test of the response of aqueous PCB concentrations and tissue concentrations towards increasing AC dose was included.

2. Conduct regional surveys to determine the benthic species recruitment pool for Hunters Point

2.1 Define recruitment pool of benthic species, along with model and data needs

The benthic community data from the reference sites in San Francisco Bay are used to determine the recruitment pool of benthic invertebrates for the Hunters Point field site. The dominant physical habitat characteristics are water depth (and therefore temperature and tidal exposure), sediment grain size, and sphere of hydrodynamic connection (e.g., ability for Hunters Point to receive a species from a reference site). These characteristics were considered for the project's sampling design. Larvae can be transported to the study area from the immediate areas north and south of the field area and from areas to the north of the Golden Gate due to the phase lag in the tidal wave between the northern and southern bay. Thus larvae at the presented experimental site may originate from benthic

communities that range from just north of Richardson Bay (Figure 5) to areas south of South Basin.

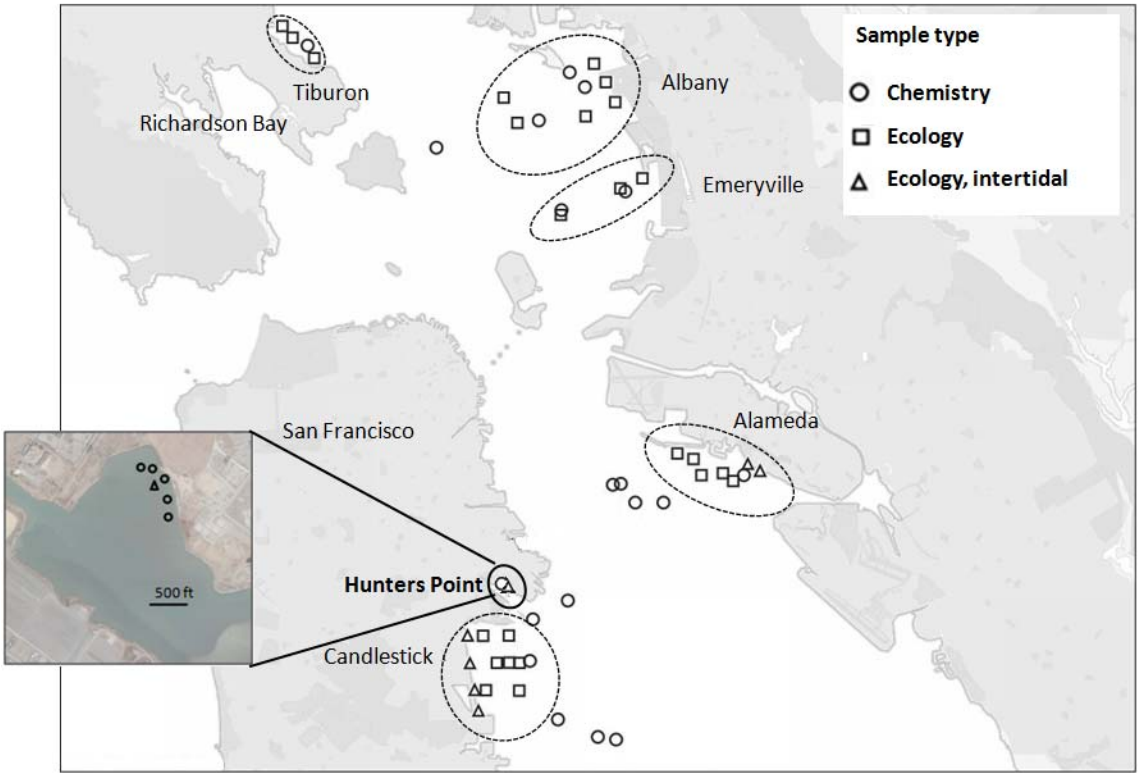

Figure 5. Benthic survey sample sites (squares) including intertidal benthic sites (triangles) and sediment chemistry sites (circled) in the Central San Francisco Bay and at Hunters Point, regional biomes are circled.

Six subsamples were collected for benthic analysis at Hunters Point. In order to take replicate samples for the conditions at Hunters Point, additional sites in the Central Bay with pollution levels similar to Hunters Point would have to be analyzed to obtain true replicates. Such sites are not present in the Central Bay. Of the six subsamples collected, three subsamples were analyzed because the benthic community results were consistent for those samples.

In addition, the benthic community was sampled following the spring and fall recruitment periods at reference stations. The selection criteria for the reference stations were based on the physical habitat to match the condition at Hunters Point most closely but with minimal pollution of the sediment. Thus, the stations were located in the Central San Francisco Bay (salinity around 30 ppt), where the sediment consists of an average of 70-80% fines with a TOC of about 1%. Few locations of muddy high-intertidal habitat are present in the Central San Francisco Bay and the intertidal reference sites chosen for the present study represent a significant portion of this habitat. To evaluate possible influences of these high-intertidal reference sites on the benthic community, the data from these sites were analyzed separate from the other reference sites.

Forty-eight reference stations (treated as replicates of reference conditions) were sampled in April 2007 and 55 stations were sampled in August 2007, April 2008, and September 2008. The reference areas were also grouped into biomes based on geographical proximity to evaluate spatial variability (Figure 5, circled areas). Six shallow intertidal stations were added to the sampling regime in August 2007. More details about the locations of the benthic stations can be found in Appendix B (item 1).

Samples were collected with a 0.05 m^2 van Veen sampler, sieved through a 0.5 mm screen, preserved in 10% buffered formalin and transferred to 70% ethyl alcohol after 1 to 2 weeks of preservation. Common organisms were identified to the species level if possible. All sorting and taxonomic work was performed by Susan McCormick (Georgetown, California). Samples were double sorted and voucher specimens maintained for future reference.

The samples were strategically processed to best identify the benthic community composition for each reference region with the financial resources available. The references regions are presented in Figure 5 (circled areas): Hunters Point, Candlestick, Alameda, Emeryville, Richmond (Albany), and Tiburon. As many stations within each region were processed until the composition of the community, as defined by the relative number of individuals in each functional group, was consistent. For example, twelve stations were sampled in the Richmond/Albany region (Appendix B1, station I.D. B5 – B16 and Figure 5) but from those the taxonomic analysis of six stations was sufficient to describe the benthic community composition of this region. From all 55 stations sampled within these regions, first, 25% of all stations in each region were selected and the number of animals in each functional group was assessed. Additional samples for each region were processed until the composition of the community was consistent. Accordingly, the functional makeup of each community did not change when 33-50% of the samples within each region were processed. On average, 50% of all samples were processed. For the intertidal references stations, all samples were sorted (samples from August 2007 and all of 2008).

The taxonomic analysis was designed to reduce the high cost of most detailed analyses that are necessary to appropriately apply certain benthic species indices. This part of the estuary within the Central San Francisco Bay is a system where species are often difficult to differentiate. Hence, species are often poorly described, and some are likely to be newly introduced to the system and therefore undescribed. The taxonomist was directed to stop at the definitions of family in difficult cases and frequently at genus for the most time consuming groups. While functional groups are usually consistent at this taxonomic resolution, it is not sufficient to meet the assumptions of a benthic index.

Functional groups were defined by feeding strategy, reproductive characteristics, and position in and protection from the sediment. Feeding strategy determines the exposure to contamination from food. The reproductive strategy determines potential mutagenic and teratogenic effects on offspring through parental transmission of genetic changes to larvae or to clones (animals that reproduce by fission). Reproductive strategy also considers the possibility that individuals have been transported as larvae from other,

non-contaminated sites. The position in the sediment defines the passive or diffusive exposure of the organism to the contaminated sediment. The uses of protective measures, such as tubes or shells, are examples of strategies to reduce exposure whereas burrowing animals that have tissue in contact with constantly changing sediment are examples of animals with maximum exposure.

Feeding functions were defined as follows: subsurface carnivores (carnivores that feed on subsurface and some surface organisms); subsurface deposit feeders (species that feed by directly ingesting the sediment below the surface); surface deposit feeder (species that feed on organic matter that either grows on the surface such as benthic algae or is transported to the surface from the water column); surface carnivore (carnivores that do not burrow below the surface of the sediment for their prey); filter and surface feeder combined (species that actively or passively remove particles from the water column and have the ability to harvest surface particles); and those that only filter feed.

Reproductive method was defined as follows: species that lay their eggs on the sediment surface; species that brood their young and release fully functional juveniles or produce clones by fission; those species that broadcast their sperm and eggs and produce pelagic larvae.

Position and protection from the sediment included the following: tubed with tissue (bare animals with sediment tubes that usually have a mucous protienaceous lining); species with similar tubes but with a chitin armour on their body; species with a chitin barrier and free living (species with a chitin armour but free-living on the surface); species with a shell barrier (calcium carbonate barrier); species with a cuticle covering (limited to nematodes); and species that expose their tissue directly to the environment and are free-living.

A detailed table of organisms and respective functional categories can be found in Appendix B.

3. Biodynamic modeling to predict PCB uptake by benthic organisms

3.1 Conduct laboratory experiments to define physiological coefficients for biodynamic model

A detailed description of the test setup for *Macoma balthica*, *Cobicula fluminea*, and *Neanthes arenaceodentata* can be found in McLeod et al. (2007) and Janssen et al. (2010), respectively (Appendix A). Physiological parameters and PCB uptake from Hunters Point sediment before and after AC amendment were measured for these organisms.

Clam bioaccumulation. For the 28-day bioassay with clams, the ingestion rate, filtration rate and PCB assimilation efficiency from the activated carbon was obtained from the literature while the PCB assimilation efficiency from sediment was measured with feeding studies using algae labeled with PCB-52 [(McLeod, Van Den Heuvel-Greve et al. 2004; McLeod, Van den Heuvel-Greve et al. 2007), Appendix A]. Briefly, sediment-sorbent contact and bioaccumulation tests were performed following previously

described procedures (McLeod, Van den Heuvel-Greve et al. 2007). Twenty clams for each replicate measure were placed atop the sediment and exposed for 28 days. Clams were fed 5 to 7 times weekly and the water in the exposure aquaria was replaced weekly. The exposure tests were conducted with untreated and AC-amended sediment. On the 28th day of exposure, clams were retrieved by hand, and rinsed with deionized water to remove sediment particles. The groups of exposed clams were each placed in water to depurate for 2 to 3 d. Clams were dissected by discarding the shells and soft tissues was combined freeze-dried, homogenized and analyzed for PCBs using hexane:acetone extraction, clean-up, and analysis with GC-ECD as described elsewhere (McLeod, Van den Heuvel-Greve et al. 2007).

Figure 6. Picture of polychaete exposure setup to measure PCB uptake rates from sediment (left) and aqueous exposure studies with clams (middle) and polychaetes (right).

Polychaete bioaccumulation. For the time-series exposure studies with polychaete, the uptake rate constant from water, uptake rate constant from sediment, ingestion rate, elimination rate and growth rate were measured. Exposure to sediment was measured weekly over 28 days with untreated Hunters Point sediment and AC-amended sediment. PCB uptake from the aqueous phase was measured over nine hours separately to differentiate passive assimilation from water and uptake of PCB by active feeding. The elimination rate was measured over 28 days in silica sand with polychaetes previously loaded with PCB by exposure to contaminated sediment for 28 days. The ingestion rate was measure with targeted feeding studies using algae labeled enriched stable isotopes. The assimilation efficiencies were estimated with the biodynamic model using the measured bioaccumulation over 28 days. Detailed descriptions of the experimental designs can be found in Appendix A.

3.2 Demonstrate applicability of biodynamic model in the field

Sediment preparation and analysis. Sediment was collected at South Basin, Hunters Point, San Francisco Bay, from a control plot (Plot B (Cho, Smithenry et al. 2007)) in January 2009, sieved (2.35 mm) and homogenized in the laboratory. About five kilograms wet sediment were amended with 3.4% activated carbon on dry mass basis (AC, TOG®-NDS 50×200, Calgon Carbon, Catlettsburg, KY, USA) and mixed on a roller for 21 days in accordance with the procedure described by Zimmermann et al. (Zimmerman, Ghosh et al. 2004). The carbon type and grain size were identical with that used in a previous in-situ study at Hunters Point (Cho, Smithenry et al. 2007).

Field bioassays. Six-week old *Neanthes arenaceodentata* were used for parallel in-situ and ex-situ bioassays. In-situ exposure cages were built following the design of Burton et al. (Burton, Greenberg et al. 2005) with a few modifications. Eight cages were prepared per test sediment (Hunters Point untreated, Hunters Point with AC-amendment) for each deployment. Each of the other three cages per test sediment were equipped with one polyoxymethylene (POM, thickness 17 μm) sampler placed in the subsurface of sediment (3 cm depth). Additional POM samplers were placed in the surface sediment layer (0.5 cm depth) for the summer (July) field deployment at Hunters Point to evaluate the availability of PCBs at the surface (Figure 7). The POM samplers were impregnated with five performance reference compounds (PRCs) to document the progression towards equilibrium. The selected PRCs were employed to represent the predominant homologs (94%) present in the field contaminated sediments; 2,4,5-trichlorobiphenyl (PCB 29) and 2,3′,4,6-tetrachlorobiphenyl (PCB 69), and 2,2',4,5',6-pentachlorobiphenyl (PCB 103), and 2,2',4,4',6,6'-hexachlorobiphenyl (PCB 155) and 2,3,3',4,5,5',6-heptachlorobiphenyl (PCB 192). Impregnation of PRCs was completed as described by Tomaszewski and Luthy (Tomaszewski and Luthy 2008).

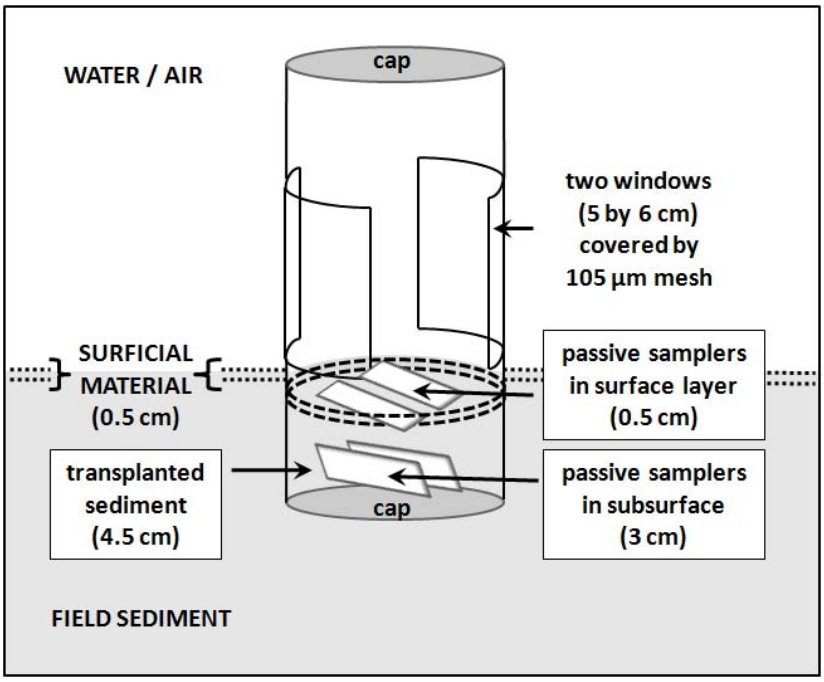

Figure 7. Schematic of field-deployed cages containing transplanted sediment and passive samplers in the surface layer (0.5 cm) and subsurface sediment (3 cm). The sediment surface in the cages lines up with the sediment surface of the field sediment.

The in-situ tests were performed in February and July 2009 at South Basin, Hunters Point. The cages were prepared in the laboratory and secured in stainless steel mesh baskets and deployed in the field. The cages with untreated sediment were deployed at the location where the sediment had been previously collected. The cages with the AC-amended sediment were deployed in an adjacent test plot (within 10 meters distance) where an in-situ trial of AC-amendment took place in 2004 (Plot A, 34.4 m^2 (Cho, Smithenry et al. 2007)).

Figure 8. Pictures showing deployment of bioassay cages in the field.

The cages were retrieved after 14 days. The organisms were removed from the sediment by gentle sieving. The individual organisms were kept in 31‰ AMW until gut clearance was completed (12 to 24 h, visual inspection). The POM samplers were retrieved and gently cleaned with de-ionized water and wiped dry with Kimwipe to remove any sediment residue. All samples were stored at -20 °C before further analysis.

Laboratory bioassay. The laboratory bioassays were designed to match the in-situ bioassay setup using a modified protocol by Millward et al. (Millward, Bridges et al. 2005). After 14 days exposure the organisms were retrieved by gentle sieving and analyzed after depuration.

Temperature probes (iBTag, Alpha Mach Inc., Mont-St-Hilaire, QC, Canada) were used to record the temperature profile of the sediment during in-situ and ex-situ tests in the sediment (3 cm depth). The average sediment temperatures in the field were 11 ± 1 ºC and 19 ± 2 ºC in February and July, respectively. Ex-situ temperatures of 21 ± 0.3 ºC were comparable to the summer field deployment [(Janssen, Oen et al. 2010), submitted to *Environmental Toxicology and Chemistry*, May 2010].

Tissue and POM analysis. Before the PCB tissue analysis, individual organism wet weight was recorded (Sartorius Micro-balance, CP2Pl) to monitor growth (N = 300). Weight measurements outside the range of ± 2.7 standard deviations were not considered for the growth rate analysis. Composite samples then were analyzed for PCBs (Janssen, Croteau et al. 2010). The organisms' lipid content was measured using individuals removed from each composite of the bioassay studies before PCB analysis using a spectrophotometic method described by van Handel (Handel 1985). PCB concentrations in POM were analyzed based on the method described by Tomaszewski and Luthy for polyethylene (Tomaszewski and Luthy 2008). Sequential extractions of the POM samplers found no significant PCBs remaining after the primary extraction.

Biodynamic modeling. A biodynamic model was used to better understand the uptake mechanisms of PCB in this study. The model as presented in equation 1 was solved for uptake rate constant from sediment, k_s (µg PCB /g ingested sediment per day) where

$$k_s = \left[C_{org}(t) - k_w \cdot C_w \cdot (1 - e^{-(k_e + k_g)\cdot t}) - C_{org}(t=0) \cdot e^{-(k_e + k_g)\cdot t}\right] \cdot \frac{(k_e + k_g)}{(1 - e^{-(k_e + k_g)\cdot t})} \cdot (C_s)^{-1} \qquad \textbf{(7)}$$

with $C_{org}(t)$ the PCB tissue concentration in the organism (µg/g dry tissue) at time t; C_s the PCB concentration of the bulk sediment (µg / g dry weight); k_w the aqueous uptake rate constant (L/g per day); C_w the aqueous PCB concentration (µg/L); k_e the rate constant of loss (1/d); k_g the growth rate constant (1/d); and $C_{org}(t=0)$ the initial PCB tissue concentration before exposure (µg/g dry weight).

The uptake rate constant from sediment, k_s, represents the product of assimilation efficiency, AE [-] and ingestion rate, IR [g sediment/g dry weight per day] where

$$k_s = IR \cdot AE \quad \textbf{(8)}$$

With a total organic carbon (TOC) content of 0.7 to 1.1% in the Hunters Point sediment, the average ex-situ IR was estimated as 8.7 gram sediment per gram dry weight of the organisms per day (Supporting Information, (Janssen, Croteau et al. 2010)). Equation 8 was used to estimate the in-situ values of IR and AE. As discussed in this study, organisms in cages with AC-amendment were exposed to a mixed diet of surficial material similar to untreated sediment and underlying treated sediment. The biodynamic model was used to estimate the apparent in-situ assimilation efficiency ($AE_{apparent}$). A mass balance approach was then used to estimate the relative contribution (x) of the AC-amended sediment as

$$\mathrm{AE}_{\mathrm{apparent}} = x \cdot AE_{\mathrm{AC-amendment}} + (1-x) \cdot AE_{\mathrm{untreated\,sedimet}} \quad \textbf{(9)}$$

using the assimilation efficiency from the AC-amendment ($AE_{AC\text{-}amendment}$) and the assimilation efficiency from untreated sediment ($AE_{untreated\ sediment}$). This expression is used to infer the extent to which the polychaetes feed on surficial sediment deposits that lack AC versus feeding on the underlying sediment with AC-amendment.

4. Development and application of a predictive general ecosystem recovery model'

4.1 Predict ecological recovery and compare to field studies

Sediment chemistry. Information about contamination levels in the surface sediment (top 5 cm) were selected from the data base of the San Francisco Estuary Institute (SFEI) and the Validation Study for South Basin at Hunters Point (SFEI ; Battelle 2004). Stations with sediment chemistry information were selected to be in the proximity of the benthic sampling regions (compare biomes in Figure 5). The values for the sediment chemistry are average values of each station, including all sampling dates available from the RMP data base. The selection of reference sites was not designed to capture a gradient in pollution, rather than replicate sites with similarly low pollution. The comparison of sediment chemistry here serves as a confirmation that the benthic reference sites are in areas with low chemical impact. The reference stations were selected to reflect the background benthic community composition of habitats similar to Hunters Point that represent the recruitment pool for Hunters Point upon successful cleanup. The chemical stations within the recruitment pool represent the pollution levels

and physical habitat conditions of this recruitment pool. Some chemical stations are not in closest proximity to the benthic stations (compare Figure 5). However, little variation has been found among the stations in the reference areas, which supports that the site selection for the benthic samples were appropriately located in low-polluted, physically similar areas. Average pollutant concentrations across reference sites were compared to Hunters Point. Detailed information about the location of the sampling sites can be found in Figure 5 and in Appendix B.

The sediment concentrations of the predominant contaminants present in the San Francisco Bay were considered in this study. These are: total PCBs, heavy molecular weight polyaromatic hydrocarbons (HPAHs), light molecular weight PAHs (LPHAs), total dichloro-diphenyl-trichloroethanes (DDTs), dieldrin, copper, lead, arsenic, mercury, and nickel; and physical properties (TOC content, fraction of fines (< 0.0625 mm), salinity). Total contaminant concentrations in the sediments were compared to evaluate if the reference sites represent locations in the San Francisco Bay that experience minimal chemical stress. Furthermore, the contaminant concentrations were analyzed to demonstrate increased chemical stress at Hunters Point relative to the reference sites. These comparisons are necessary to justify the selection of study sites being appropriate to evaluate PCB-induced changes of the benthic community composition.

The contaminant levels were further compared to the effective range low (ERL) and effective range medium (ERM) as suggested by the sediment quality guidelines (SQG) (Long, MacDonald et al. 1995) to assess the likelihood of ecological risk at these locations. The contaminant-specific ERL and ERM values suggest threshold concentrations in sediment for the 10^{th} and 50^{th} percentile of expected adverse biological response, which are empirically derived including observed responses in bioassay and benthic community tests. The ERM quotients (ERMq) were calculated for Hunters Point and the reference sites by dividing the average sediment concentrations of each contaminant by the respective ERM value. The ERMq allows assessing the number of pollutants that exceed the SQG and the magnitude for each site. The sum of all ERMq for the contaminants considered (ERMq) was calculated and allows comparing the cumulative chemical stress at the sites. Lastly, the median ERMq as the average of all ERMq exceeding one (mERMq >1) was computed. Usually the mERMq values represent the median ERMq for all contaminants considered, including those which do not exceed the respective ERM threshold. However, the more contaminants are included that are not present or present at very low concentrations, the lower would be the mERMq even though some contaminants may be significantly exceeding the guidelines. Thus, the use of mERMq >1 emphasizes those contaminants which pose a risk and allows quantifying and comparing the cumulative risk posed by chemicals among sites.

Ecosystem Recovery. The benthic community structure has been assessed at Hunters Point and reference sites in the San Francisco Bay (see task 2) to disclose critical taxa by differences in the ecosystem structure and to identify the recruitment pool for Hunters Point.

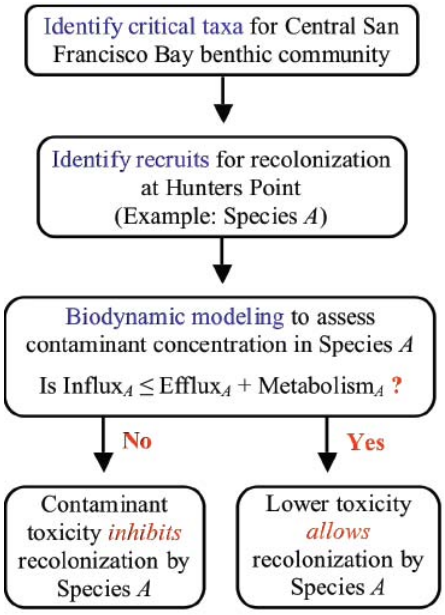

Figure 9. Flow-chart representation of ecological recovery modeling to determine the re-colonization potential of possible recruits.

The information from the benthic community surveys allow us to assess the natural variability of species and to test whether physiology-functional ecology can reveal the type of species that are adversely effected at Hunter's Point. The biodynamic model was used to estimate total PCB tissue concentrations (equation 1). Tissue concentrations were predicted for three organisms that represent different feeding strategies: the filter and surface feeding clam *Macoma balthica*, the deposit feeding polychaete *Neanthes arenaceodentata*, and the filter feeding mussel, *Mytilus edulis*, which are organisms that inhabit the San Francisco Bay. The species-and PCB-specific physiological parameters used for the model predictions are listed in Table 2. The exposure of filter feeding organisms through ingestion of particulate organic matter, POM, is variable, depending on its origin, and the concentration and composition of POM in water (Burton and Johnston 2010). The PCB concentration of the POM is assumed to be similar to the PCB concentration of the surface sediment as measured previously for Hunters Point by Cho et al [1]. It is possible that the PCB concentrations in POM can be lower than in the surface sediment when particulate matter originates from less polluted areas. The present study employs conservative estimates, where POM and surface sediment have similar PCB concentrations. Previously, the ingestion rate of particulate organic carbon, POC, was measured for *M. edulis* as 0.4 mg POC/g per hour or 0.01 g POC/g per day [2] at POC water concentrations typical for the Central San Francisco Bay with (0.5 to 1.0 x 10^{-3}g/L, [3]). The ingestion rate of POC can be converted to ingestion rate of POM, IR_{POM}, assuming that POM has the general composition of $C_5H_7O_2N$. Then POC represents 53.09% of POM by molecular weight and IR_{POM} equals about 0.02 g POM/g per day.

Various exposure conditions were tested to assess the response of PCB uptake relative to the dietary exposure pathway of the different model organisms. The exposure

scenarios further allow comparing the internal PCB concentrations at Hunters Point to exposure conditions with different PCB water and sediment concentrations.

Table 1. PCB sediment concentrations.

Location	*PCB concentration in sediment [ppb]*
Hunters Point	1570[a]
Oakland Harbor (hot spot)	476[b]
Clean-up goal	200[c]
ERM	180[d]
ERL	23[d]
Reference sites in Central Bay	9[e]

[a](Battelle 2004); [b](McFarland, Clarke et al. 1994);
[c](Brajas 2008); [d](Long and MacDonald 1998); [e](SFEI).

The contaminant tissue concentrations were calculated for (1) the present condition at HP, (2) the conditions of the selected reference sites, (3) the Effective Range Low (ERL) for PCBs, (4) the Effective Range Median (ERM) for PCBs, (5) the PCB cleanup goal for HP, and (6) the present concentrations at a site in Oakland Harbor (sample station identification IT-6 and IM-1; (McFarland, Clarke et al. 1994)). Respective PCB sediment concentrations range over two orders of magnitude (Table 1)

To estimate the PCB pore water concentration under different exposure conditions, general partitioning theory was applied. The average partitioning coefficient (K_d) was estimated as the ratio of total PCB sediment concentrations and pore water concentrations for Hunters Point as 1.01×10^5 L/kg averaging various measurements for the site (Table 3). The value of K_d is sediment–specific and highly depended on the sediment TOC content and fraction of fines. Pore water concentration or K_d value were not available for the reference sites. To estimate the pore water concentrations for further modeling, the values were estimated using the Hunters Point average K_d value. The K_d values at HP range from 3.59×10^4 to 1.89×10^5 L/kg and it was assumed that the K_d values at the reference sites are within this range because the TOC content and fraction of fines is similar amongst all sites. The PCB concentration in the overlying water is about one order of magnitude lower than the pore water concentration (Cho, Smithenry et al. 2007).

For the Oakland Harbor sediment a comparable TOC content relative to Hunters Point has been reported but the fraction of fines is very low [silt plus clay equals 23%, (McFarland, Clarke et al. 1994)]. Nevertheless, the PCB hot spot at Oakland Harbor is used as a demonstration site representing significantly elevated PCB sediment concentrations relative to the reference conditions in the Central Bay but significantly lower than at Hunters Point.

Biodynamic modeling was further used to estimate the PCB tissue concentrations expected at Hunters Point after the AC-amendment. Information about the relative reduction of PCB availability after AC addition was obtained for the three test species from the literature (Table 2). The expected remedial response was estimated for favorable field and treatment conditions, i.e., the site should be depositional, it should allow for mixing AC into the upper sediment layer, and it should be dominated by the fast desorbing fraction (non black carbon-like particles).

Table 2. Values of physiological and model parameters for the benthic invertebrates.

Parameter, symbol and unit	*Neanthes arenaceodentata*	*Macoma balthica*	*Mytilus edulis*
Filtration rate, FR, L water/ g dry wt / d		2[c]	45[d]
Aqueous assimilation efficiency, AE_{aq}, %		50[c]	20[e]
Aqueous uptake rate constant, k_w, L water / g dry wt / d, = FR x AE_{aq}	0.5[a]	1	9
Ingestion rate, IR, g sediment / g dry wt / d	3.5[b]	0.25[c]	0.02[h]
Sediment assimilation efficiency, AE_s, %	7[b]	20[c]	10[i]
Elimination rate constant, k_e, 1 / d	0.04[a]	0.05[c]	0.144[e]
Growth rate constant, k_g, 1 / d	0[b]		0.002[f]
Expected time to attain steady state based on model predictions, d	56	100[c]	40
lipid content of dry weight,[*] %	5[a,b]	18[g]	10[f]
remedial success of AC, given as % reduction of bioaccumulation	90[b]	84[c]	90[d]

[a](Janssen, Croteau et al. 2010), [b](Janssen, Oen et al. 2010), [c](McLeod, Luoma et al. 2008), [d] (Tomaszewski, McLeod et al. 2008), [e](Bj rk and Gilek 1997), [f](Gilek, Bj rk et al. 1996), [g](Wenne and Polak 1989), [h]estimated as described above, [i](Bj rk and Gilek 1999), [*]with the dry wt of the *N. arenaceodentata* being 10% of the wet weight.

Finally, the model was used to approximate the reduction of PCB availability required to achieve tissue concentrations at Hunters Point that are comparable to tissue concentrations expected under other exposure scenarios (reference sites, SQG thresholds ERL and ERM, Oakland Harbor, cleanup goal). The AC-amendment reduces the assimilation efficiency of PCB from sediment (AE_{sed}) as well as the aqueous PCB concentration (C_w). The required reductions in PCB availability is the fraction by which both parameters (AE_{sed} and C_w) have to be reduced, while keeping the total PCB sediment concentration at Hunters Point constant (C_{sed} = 1570 ppb).

These estimates were compared to the expected remedial response observed in our bioassays to evaluate the potential of ecosystem recovery at Hunters Point.

Table 3. Measured total PCB sediment and pore water concentrations and computed K_d values for Hunters Point.

C_S [mg/kg]	C_w [mg/L]	K_d [-]	
7.0	3.70×10^{-5}	1.89×10^{5}	d
6.5	5.20×10^{-5}	1.25×10^{5}	d
3.0	3.75×10^{-5}	8.00×10^{4}	a
2.0	3.70×10^{-5}	5.41×10^{4}	b
1.1	8.00×10^{-6}	1.38×10^{5}	d
1.7	9.70×10^{-6}	1.75×10^{5}	d
1.0	8.70×10^{-6}	1.15×10^{5}	d
0.8	2.23×10^{-5}	3.59×10^{4}	d
1.5	2.45×10^{-5}	6.12×10^{4}	d
1.8	2.35×10^{-5}	7.66×10^{4}	d
2.2	2.16×10^{-5}	1.02×10^{5}	d
1.8	3.13×10^{-5}	5.75×10^{4}	d
1.2	1.11×10^{-5}	1.08×10^{5}	d
1.4	8.90×10^{-6}	1.57×10^{5}	d
1.0	9.00×10^{-6}	1.11×10^{5}	d
1.5	1.45×10^{-5}	1.03×10^{5}	d
1.3	1.31×10^{-5}	9.92×10^{4}	d
1.0	1.03×10^{-5}	9.71×10^{4}	d
1.1	1.37×10^{-5}	8.03×10^{4}	d
0.8	6.50×10^{-6}	1.23×10^{5}	d
1.1	7.10×10^{-6}	1.55×10^{5}	d
0.8	1.68×10^{-5}	4.76×10^{4}	d

[a] (McLeod, Luoma et al. 2008), [b](Janssen, Croteau et al. 2010), [c](Tomaszewski and Luthy 2008), [d] unpublished data

4.2 Correlate conventional alternative assessment methods and model ecosystem recovery in the field

The present study compared the trait-based risk assessment with alternative measures, where appropriate: total abundance and occurrence of dominant species. The Shannon-Wiener Diversity Index with its standard error (Jackknife technique with n = 10,000, Pisces Conservation 2007 Software) was computed, although there is some reservations about using this index with the study design because the taxonomic resolution for some rare benthic species was not always sufficient to meet the assumptions of a benthic index (compare task 2). Furthermore, benthic communities at all locations were compared using non-parametric multi-dimensional scaling (MDS) with PRIMER 6 (Clarke 1993; Clark and Warwick 2001). Abundance data for all species were square-root transformed and Bray-Curtis similarities were computed between each pair of stations. The resulting matrix was ordinated by non-metric multi-dimensional scaling to display the variation among the assemblages.

5. Monitor and assess carbon amendment in Grasse River

5.1 Bioaccumulation studies with Lumbriculus variegates

Laboratory bioaccumulation experiments were conducted to study PCB bio-uptake by *L. variegatus*. Cultures of *L. variegatus* were purchased from Aquatic Foods, Inc. (Fresno, CA, USA) two weeks prior to the start of the experiment. Worms were maintained, unfed, in aerated aquaria with 1 to 2 cm sand until use. Immediately prior to the start of each experiment, worms were removed using disposable pipettes and transferred to polystyrene weigh boats in groups of twenty individuals. Selected worms were 2-3 cm in length to minimize effects of growth and/or reproduction. At the start of each experiment, 20 worms each were placed in each exposure beaker.

Semi-static bioaccumulation experiments were performed in 100 mL glass beakers following U.S. EPA protocols (USEPS 2000). Beakers contained 80 g sediment and 15-20 mL synthetic freshwater. Sediment was added to the beakers three days before the introduction of worms, and the overlying water was replaced daily during that period. Worms were exposed to untreated ($\sum_{PCBs}$ = 13.5 mg/kg dry wt) Grasse River sediment, and Grasse River sediment that was pre-treated with activated carbon (TOG® 50x200; Calgon Corporation, Pittsburgh, PA, USA) at 2.5% (dry wt) for 30 days under well-mixed conditions. The 28-day exposure tests were performed with the oligocheates and additional exposure test were done for various time-points throughout the uptake period.

Additional worms were exposed for 28 days, then allowed to depurate in clean sand for one to seven days, with worm beakers sacrificed in triplicate at various time-points throughout the depuration period. Following sacrifice, worms were allowed to depurate for six hours. Following this time, they were frozen, and then freeze dried. Freeze dried tissues were extracted using sonication with hexane:acetone. Extracts were cleaned up using silica gel columns. PCB concentrations were measured via GC-ECD. Aqueous PCB concentrations were obtained using polyethylene devices (PEDs) as passive samplers. Polyethylene (50 µm diameter; 0.92 g/cm^3) was purchased from Brentwood Plastics, and was the same as used in the Hunters Point passive sampling work. Aqueous PCB concentrations were measured after 28 days of mixing in 40-mL vials.

Results were modeled using the biodynamic model explained earlier (equation 1). For PCB uptake by *L. variegatus*, the aqueous route of exposure has been shown to be negligible. Therefore the uptake was modeled using the biodynamic equation with sediment uptake only.

5.2 PCB mass transfer modeling with heterogeneous mixing

Site Description. The demonstration test site is a shallow tidal mudflat in South Basin adjacent to Hunters Point Shipyard, San Francisco, CA, USA (Figure 10) with water depth ranging from 1.8 m to less than 0.6 m (Brajas 2008). This site is exposed briefly at low tide, emerging only during the lowest stage of the lower low water events of this mixed sediment-intertidal system. Historical site activities at the shipyard resulted in the release of chemicals to the environment, including offshore sediments (Battelle

2004). There exists no visible surface water channel on the test sites. A sheet pile wall and rip-rap exist along the shoreline to prevent possible contamination from the upland. The combined results of Sedflume experiment (Zimmerman, Bricker et al. 2008) with site sediment and comprehensive hydrodynamic modeling studies (Brajas 2008) indicate that the South Basin area is a net depositional zone. The site comprises cohesive sediments not subject to exceeding sediment critical shear stress in most storm events (Zimmerman, Bricker et al. 2008). The top 10 centimeters of the sediment in the demonstration area is composed of small gravel, shells, and clay particles. Underneath this top layer is a more homogenous layer of clay, characteristic of bay mud. The sediment has a mid-range PCB concentration of 1-10 ppm (Battelle 2004). Because PCBs tend to adsorb to fine-grained sediment particles and organic matter, sediment resuspension and deposition are major contaminant transport pathways in South Basin. The net sediment deposition rate is about 1 centimeter per year or less (Battelle 2004).

Two of four test plots were utilized that were assessed in the previous field-scale AC application project (Cho, Ghosh et al. 2009), identified as Plots D and E (Figure 10,(C)). The two plots were separated by a distance of 4 m and were located approximately 25 m from the shoreline, within the tidal mudflat region. In Plot D, about 2 wt% of activated carbon (AC) was mechanically mixed down to a nominal 30 cm depth in January 2006, while Plot E served as an unmixed reference plot. Detailed descriptions of the test site, test plots, and AC introduction are given by Cho et al. (2009).

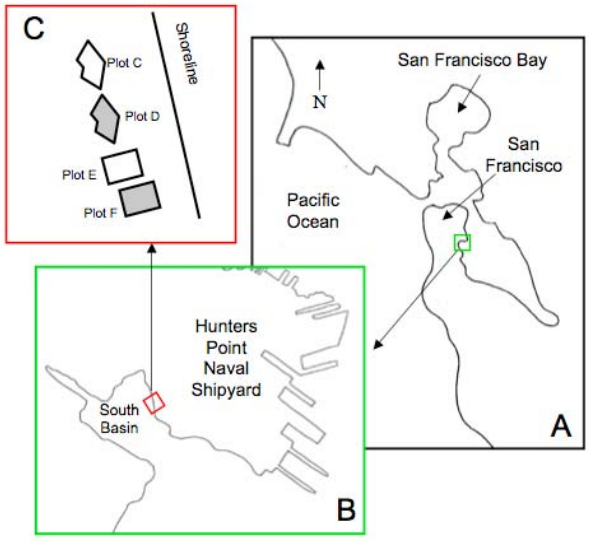

Figure 10. Schematic of (A) San Francisco Bay; (B) Hunters Point Naval Shipyard and South Basin; and (C) four test plots used in the field-scale study of *in-situ* AC amendment [Cho et al., 2009]. The two plots indicated by shading (Plots D and F) were treated by mixing the sediment with AC to a nominal 30-cm depth. Plot C served as a control plot, and Plot E served as an unmixed reference plot. In this study, AC treated plot D and unmixed reference plot E were used for assessment of pore water movement.

Vertical Sediment Temperature Profile. Vertical sediment temperature profiles were measured for 14 days in mid-August 2007 and early March 2008. The sampling method was similar to the method described by Cho et al. (2005). At each sampling time, two temperature-logging stations were installed at the center of each plot (Figure 11(A)), separated by approximately 0.6 m. After 14 days of data logging, the stations were retrieved and data downloaded. Each temperature logging station had eight temperature loggers (iBCod Type 22L, Alpha Mach Inc.) encapsulated in polyurethane casting to prevent corrosion (Figure 12). The probe has a hole in one side where the temperature sensor is exposed to the surrounding environment. The outer length is about 4.5 centimeters. The temperature loggers had a resolution of 0.0625 ºC, with a precision of 0.5 ºC. A total of eight temperature probes are mounted to a hardwood rod to make one temperature logging station. These were situated at two levels above (15 cm, 2.5 cm) and at six levels below (2.5 cm, 5 cm, 15 cm, 30 cm, 45 cm, and 60 cm) the sediment surface. At the beginning of each sampling period (August 2007 and March 2008), temperature-logging stations were installed at the test site during low tide. Each station was pounded into the sediment, and visually ensured to have a tight seal with sediment layer. The strong seals between the temperature logging stations and sediment layer were checked again when the stations were retrieved manually.

During temperature sampling periods, incoming shortwave solar radiation was measured by a solar radiation sensor (Solar Radiation Sensor and Micro Station Data Logger, Onset Computer Corporation) that was installed at the shore (Figure 11(B)). The tide predictions by the National Oceanic and Atmospheric Administration (NOAA) were used as sea level data, which was corrected to use the surface elevation of the test plots (0.5 m above sea level) as the new datum (http://tidesandcurrents.noaa.gov).

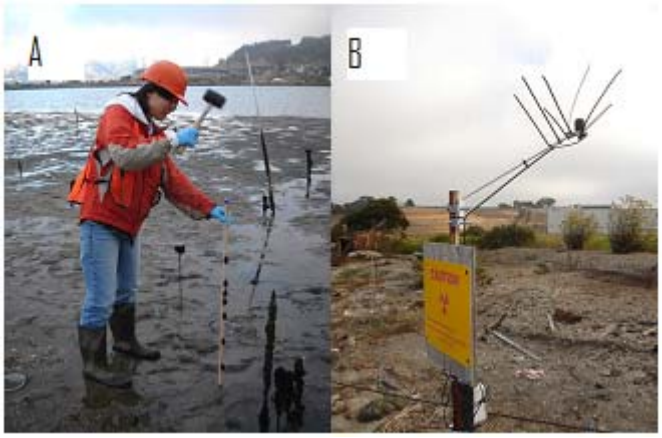

Figure 11. (A) Deploying temperature observation stations at the center of each plot; (B) The mico-weather station installed at the shore to measure solar radiation.

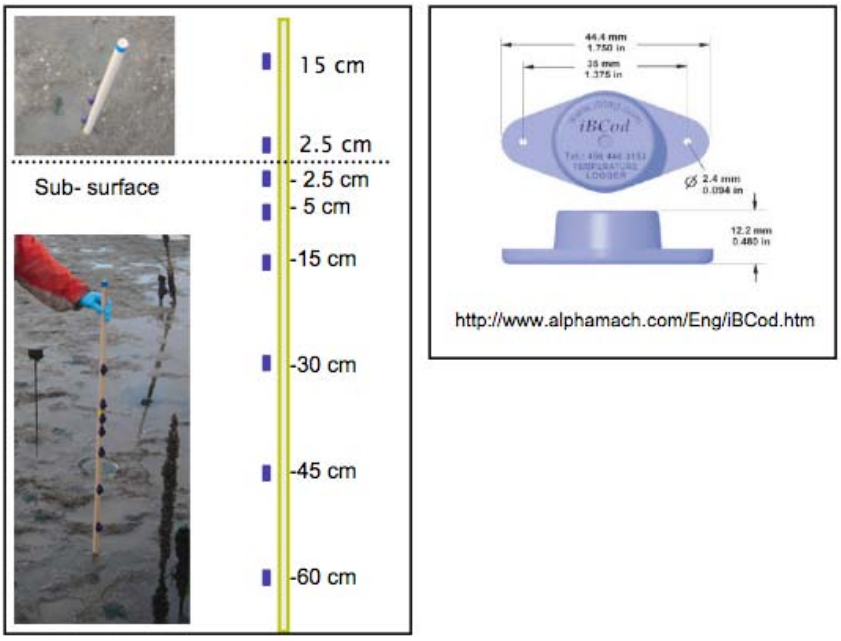

Figure 12. Left: schematic and pictures of a temperature logging station; Right: picture of a temperature probe.

Modeling

Heat Transport Model

A heat transport model was developed to assess the extent of heat transport by advective pore water movement in the saturated mudflat. The derivation and implementation of the heat transport equations followed the method developed for the commercial numerical simulation program SHEMAT for reactive groundwater flow (Clauser 2003). Assuming no heat sinks or sources in the sediment layer, the change in volumetric heat storage is equal to the sum of heat transport by diffusion and advection. For one dimensional heat flow in the vertical direction (z),

$$\frac{\partial(\rho_b c_b T)}{\partial t} = \frac{\partial}{\partial z}(k_b \frac{\partial T}{\partial z} - \rho_w c_w T v) \quad \textbf{(10)}$$

where, ρ_b is the bulk density of the system (kg m^{-3}), c_b is the bulk specific heat capacity (J kg^{-1} K^{-1}), T is the temperature (K), t is the time (s), k_b is the bulk thermal conductivity (W m^{-1} K^{-1}), ρ_ω is the water density (kg m^{-3}), c_w is the water specific heat capacity (J kg^{-1} K^{-1}), v is the Darcy velocity (m s^{-1}). The linear combinations of solid and water contributions weighted by the porosity ϕ was used to estimate the bulk heat capacity (Clauser 2003). For estimation of bulk sediment thermal conductivity, the geometric mean model was used (Woodside and Messner 1961; Goto and Matshbayashi 2009). Assuming constant material properties during the time interval and spatial homogeneity,

$$(\phi\rho_w c_w + (1-\phi)\rho_s c_s)\frac{\partial T}{\partial t} = (k_w{}^{\phi} k_s{}^{(1-\phi)})\frac{\partial^2 T}{\partial z^2} - \rho_w c_w v\frac{\partial T}{\partial z} \quad \textbf{(11)}$$

where, ρ_s is the solid density (kg m^{-3}), c_s is the solid specific heat capacity (J kg^{-1} K^{-1}), k_s is the solid thermal conductivity (W m^{-1} K^{-1}), and k_w is the water thermal conductivity (W m^{-1} K^{-1}).

Additionally, the consideration of mechanical dispersion will modify equation 11 as,

$$\begin{aligned}&(\phi\rho_w c_w + (1-\phi)\rho_s c_s)\frac{\partial T}{\partial t}\\ &= (k_w{}^{\phi} k_s{}^{(1-\phi)})\frac{\partial^2 T}{\partial z^2} + \phi\rho_w c_w D_{disp}\frac{\partial^2 T}{\partial z^2} - \rho_w c_w v\frac{\partial T}{\partial z}\end{aligned} \quad \textbf{(12)}$$

where, D_{disp} is the mechanical dispersion coefficient (m^2 s^{-1}). Equation 12 was discretized forward in time and centrally in space for numerical simulation.

Model parameters were obtained experimentally or from the literature (Table 4). Solid thermal conductivity was determined by *ex-situ* measurements (2.27±0.27 W m^{-1} K^{-1} (Maeba 2009)). The initial temperature conditions for each simulation were from the measured data. The upper-boundary condition for simulations was the measured temperature at 2.5 cm depth. As the lower boundary condition at 60 cm, a constant temperature was assumed.

For model calibration, two simulation scenarios were considered, and the best-fit parameter was determined by the least root mean squared error (RMSE) between the simulation results and field measurements. First, the limiting case that heat transport could be explained by heat diffusion and advection without mechanical dispersion was tested. By neglecting the mechanical dispersion term, the directional advective flow was tried to be maximized. The equation for this scenario was,

$$(\phi\rho_w c_w + (1-\phi)\rho_s c_s)\frac{\partial T}{\partial t} = (k_w{}^{\phi} k_s{}^{(1-\phi)})\frac{\partial^2 T}{\partial z^2} - \rho_w c_w v\frac{\partial T}{\partial z} \quad \textbf{(13)}$$

The second limiting-case scenario assumed no advection but non-zero mechanical dispersion. This assumption was intended to consider a plausible case of a mud flat with very small net advection but with considerable mechanical dispersion. Although advective flow and mechanical dispersion coexist in a real system, our objective was to define the plausible range for possible advective flow velocities and dispersion coefficients at the field site. Therefore, it was assumed that the contribution of the dispersion term is negligible when fitting the maximum plausible vertical advective flow velocity in the first scenario and vice versa in the second scenario:

$$(\phi\rho_w c_w + (1-\phi)\rho_s c_s)\frac{\partial T}{\partial t} = (k_w{}^{\phi} k_s{}^{(1-\phi)})\frac{\partial^2 T}{\partial z^2} + \phi\rho_w c_w D_{disp}\frac{\partial^2 T}{\partial z^2} \quad \textbf{(14)}$$

There is a limit to the minimum Darcy velocity and mechanical dispersion coefficient that can be reliably derived from temperature measurements and heat transport modeling. The minimum extractable parameter can be defined by the ratio of the relative weight of each process with a tolerance factor. For a tolerance of 1%, the detection limit for the dispersion coefficient would be,

$$D_{disp,\min} = \frac{(k_w{}^{\phi} k_s^{(1-\phi)})}{\phi \rho_w c_w} \times 0.01 = 5.6 \times 10^{-9} m^2/s \quad \textbf{(15)}$$

The detection limit of Darcy velocity was indirectly calculated using the minimum detectable dispersion coefficient. Assuming dispersivity as 10 % of the length of system (0.6 m) (ASCE 1996), the minimum detectable Darcy velocity would be 0.8 cm/d.

Table 4. Parameters for the heat transport model.

Parameter	Value	Source
ϕ porosity	0.57	Brajas & Associates Inc. and Tetra Tech EM Inc., 2008
ρ_ω water density	1021.0 kg m^{-3} (20 ºC) 1022.1 kg m^{-3} (15 ºC)	(Kaye and Laby)
ρ_s solid density	2300 kg m^{-3}	Maeba, 2009
c_w water specific heat capacity	3993 J kg^{-1} K^{-1}	(Kaye and Laby)
c_s solid specific heat capacity	780 J kg^{-1} K^{-1}	Clauser Ed., 2003
k_w water thermal conductivity	0.596 W m^{-1} K^{-1}	(Kaye and Laby)
k_s solid thermal conductivity	2.27 W m^{-1} K^{-1}	Maeba, 2009

PCB Mass Transfer Model

A PCB mass transfer model is based on the model that has been developed by Werner (Werner, Ghosh et al. 2006). The MATLAB code of this model was obtained from Dr. Werner, and modified to consider site-specific parameters and possible convective flow of pore water. This work is under development and will be the subject of a future proposal.

The outline of a future mass transfer model would consider both heterogeneity of carbon distribution, possible advective flow, e.g., three cm/d, and diffusive mass transfer. Exemplary modeling scenarios are shown below.

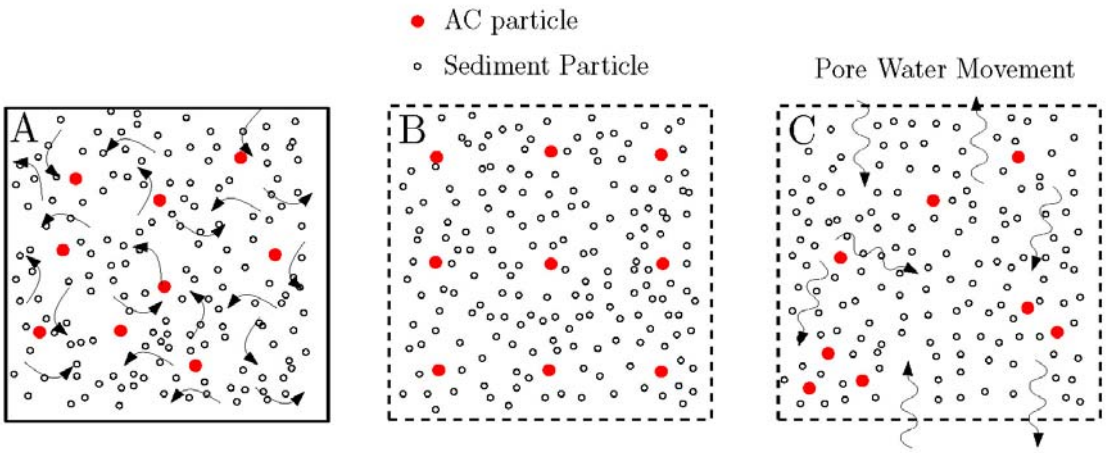

Figure 13. Illustrative activated carbon amendment and sediment mass transfer scenarios: Left, well mixed homogeneous conditions as in the laboratory; Center, homogeneous carbon distribution with diffusive mass transfer; Right, heterogeneous carbon distribution with advective pore water movement and diffusive mass transfer as may occur in the field.

V. Results and Accomplishments

1.1 Characterize (in the laboratory) relevant parameters for the use of polyethylene sampling devices

Laboratory Time Series. In the well-mixed laboratory conditions, PCBs with log K_{ow} < 6.8 achieved equilibrium, such as PCB 95 over 99 days (Figure 14). However, uptake of total PCBs increased throughout the 99-day sampling period.

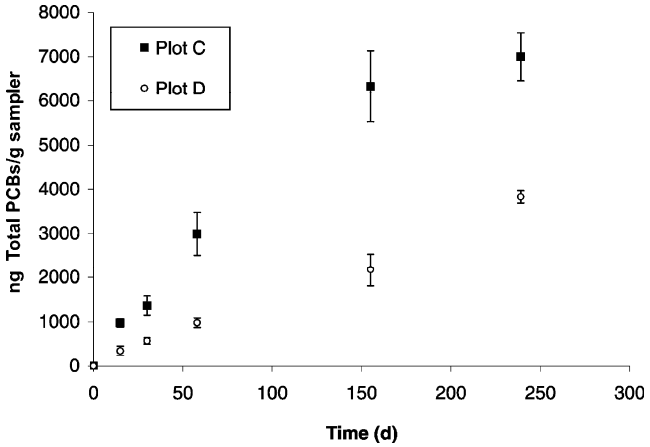

Figure 14. Uptake of PCBs over time in PEDs deployed in sediment in mixed-only Plot C (squares) and activated carbon amended Plot D (circles) at Hunters Point, San Francisco, CA (*n* = 3).

A notably larger amount of PCBs was taken up by PEDs over time as compared to the field series. Therefore, much higher exchange rates were measured in the laboratory, as the well-mixed conditions increased flow velocities in the laboratory sediment slurries.

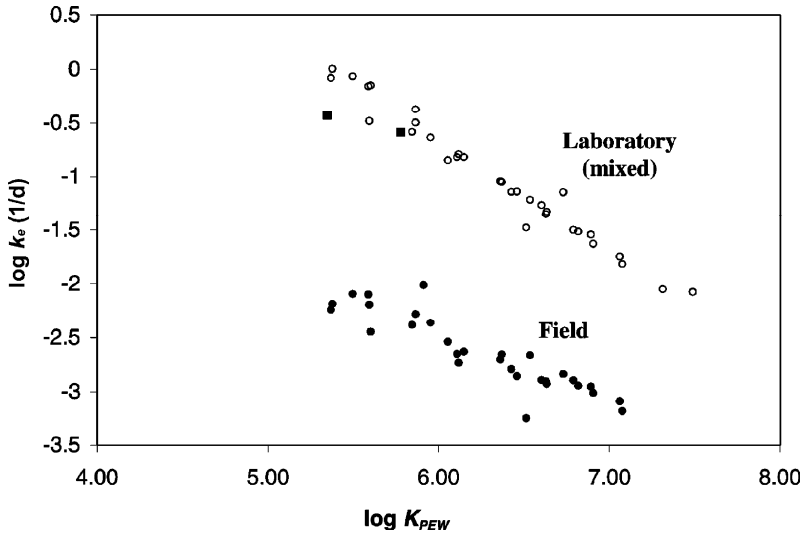

Figure 15. Exchange rate coefficients (k_e) as a function of log K_{PEW} estimated from uptake of PCBs from contaminated sediment in the field (closed circles) and in the laboratory (open circles) or dissipation of PRCs in the laboratory (squares).

The calculated results for field k_e values are shown in Figure 15. Vrana et al. (Vrana and Schuurmann 2002) comment the transport kinetics governed by the aqueous boundary layer are indicated by a decrease in *k*e values with increasing K_{ow}, as noted in Figure 15 for the field data, which validates the use of equation 2 to model uptake kinetics.

With these analyses the first milestone of the PED work was completed in 2007 [(Tomaszewski and Luthy 2008), Appendix A].

1.2 Apply PEDs and PCB immunoassay in the field at treated and untreated sites

Immunoassay analyses on sediments. Total PCB concentrations measured were shown in Table 5. Three to five sub-samples per each plot sample were extracted, measured for their PCB concentrations, and averaged.

Sampling location	**Treatment**
Plot C	mixing, no activated carbon amended
Plot D	mixing, activated carbon amended
Plot E	no mixing, no activated carbon amended
Plot F	mixing, activated carbon amended

Table 5. Sediment PCB concentrations measured by immunoassay analysis and GC.

Plot	Conc. by immunoassay [ppm][a]	Std. dev.	Conc. by GC-µECD [ppm]	Std. dev.
C (2007)	0.143	±0.087	1.058	±0.301
D	0.096	±0.003	0.845	±0.136
E	0.128	±0.049	0.896	±0.073
F	0.104	±0.011	0.839	±0.163
C (Top 6”)	0.588	±0.119	0.468	±0.163
D	0.203	±0.108	0.499	±0.566
E	0.463	±0.032	0.613	±0.056
F	0.335	±0.113	0.239	±0.008
C (Bottom 6”)	1.028	±0.167	1.019	±0.042
D	0.766	±0.285	0.531	±0.466
E	1.318	±0.149	1.414	±0.499
F	0.706	±0.215	0.410	±0.022
C (2008)	0.865	±0.373	1.496	±1.135
D	0.308	±0.041	0.356	±0.158
E	0.498	±0.041	0.672	±0.406
F	0.681	±0.170	1.031	±0.716

[a] The concentration reported as Aroclor 1254, in µg of PCB/g of sediment. The number of sub-samples ranges from three to five.

The concentrations throughout different sample sets were lower in plot D and F where AC was amended than the mixing control plot C and non-mixing control plot E. Here, the one exception is the composite sample in 2008 from plot F. However, its value is still lower than the mixing control (Plot C). The lower PCB concentrations in plot D and E were consistent with observations using conventional PCB analysis by gas chromatography with micro electron capture detector (GC-μECD) (Table 5).

Correlation between the immunoassay analysis and GC analysis. PCB concentrations measured by GC-μECD are shown in Table 4 to compare with the immunoassay test results. The total PCB concentrations reported from GC analyses were sums of individual PCB congener concentration calibrated using LMMB standards. As mentioned above, the concentrations from the immunoassay test were reported as Aroclor 1254, not the sum of the individual PCB congeners. Thus, the trend of the data, not the values, should be compared.

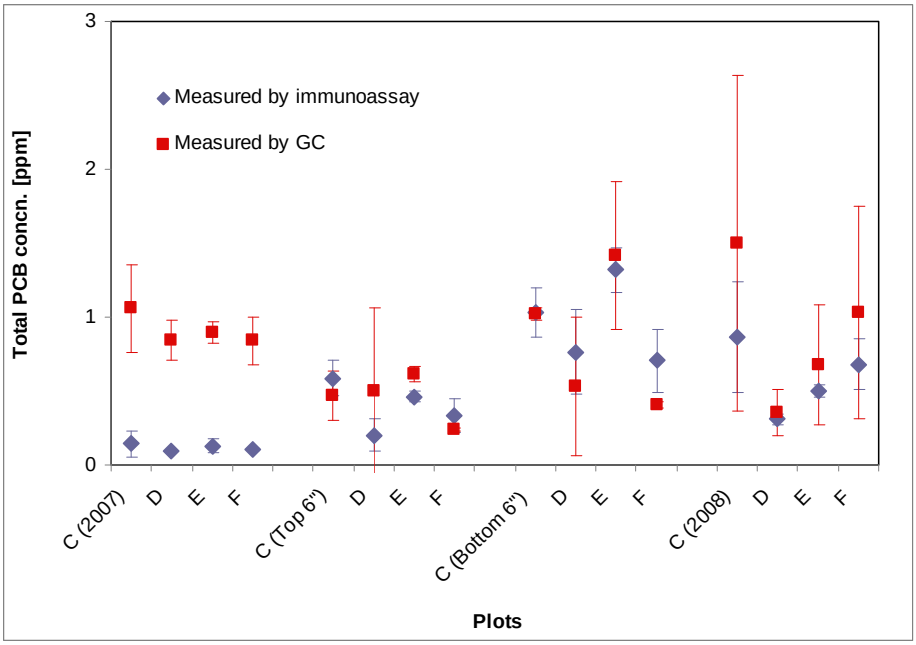

Figure 16. Comparison of the sediment PCB concentrations measured by immunoassay and GC.

Figure 16 shows that the values are in the similar pattern throughout the sample sets. To find correlations between two measurements, two data sets were compared as shown in Figure 17. The values of the square of the correlation coefficient, R^2s, range from 0.82 to 0.85 showing a general agreement except sediment samples collected in 2007 (data not shown). The correlation was better when the data were compared by each sample set than when all data from different sampling events were compared. The heterogeneity of the samples and different sensitivity of the method suggest only semi-quantitative data are obtained by the immunoassay method.

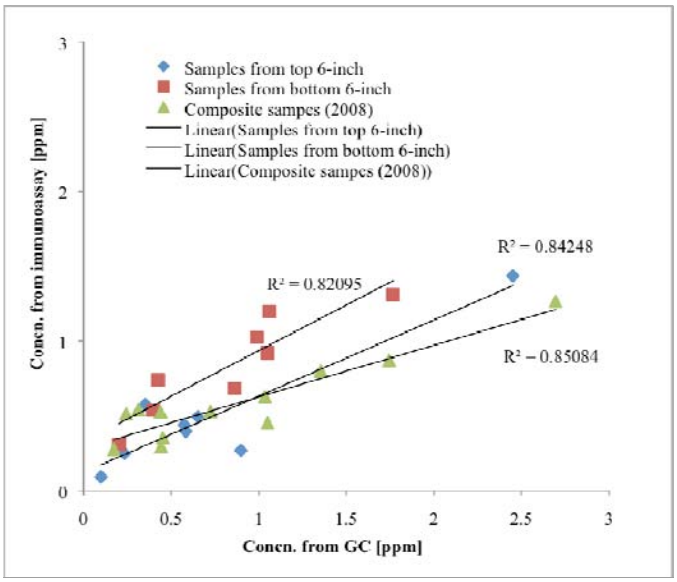

Figure 17. Correlation between individual measurements (non-averaged) of immunoassay (*n*=2-3) and GC analyses (*n*=3-5).

Correlation between the immunoassay analysis and pore water concentration measurement by PEDs. The pore water concentrations were measured from the polyethylene sampling devices, PEDs that have contacted activated carbon amended sediment in the field for 18 month. PED-measured pore water concentrations ranged from 7.14 to 17.02 ng/L. The regressions presented in Figure 17 were used to convert immunoassay values to 'GC measurement-based values' because a better correlation with pore wate concentrations was expected.

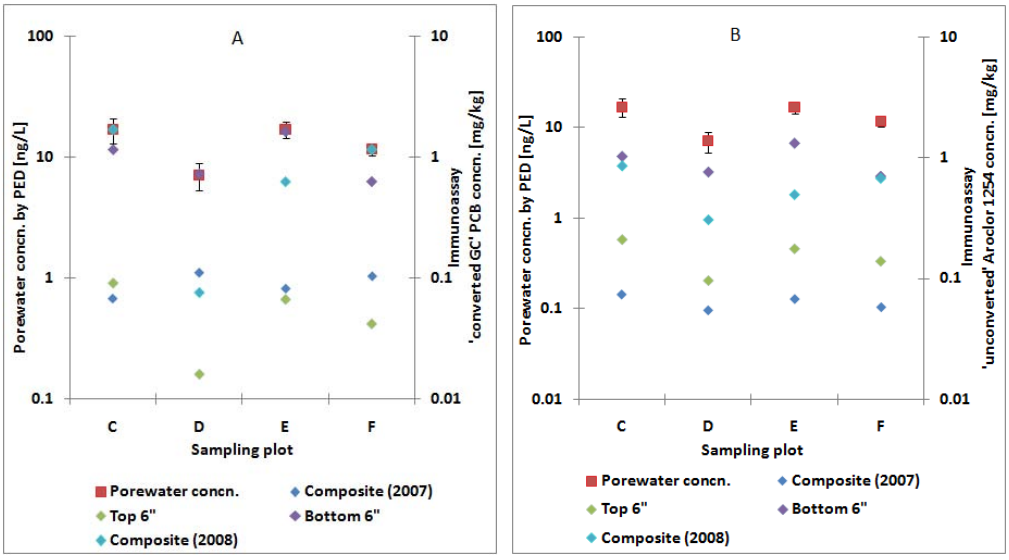

Figure 18. Pore water concentrations measured by PEDs (in red) compared to measured immunoassay concentrations converted to GC-based values (a) and unconverted Aroclor 1254 based values (b).

Data in Figure 18 shows pore water concentrations measured with PEDs and the immunoassay as converted values (a) and non-converted (direct) values (b).

The difference in the units should be noticed. Pore water concentrations were reported as ng of PCB per liter of water and PCB concentrations by the immunoassay test were expressed in mg of PCB per kg of the sediment, and the differences in the values are about five orders of magnitude (~ 10^5). This difference is reasonable considering the difference of the medium. Higher PCB concentrations are found in the sediment than in water due to the contaminants large hydrophobicity. Considering that the octanol-water partitioning coefficient, K_{ow}, of PCBs are in the range of 10^4 to 10^6, the concentrations measured by PEDs and by immunoassay may correlate once the unit of PCB concentrations were divided by K_{ow} values or K_{oc} (organic carbon-water partitioning coefficient) values of corresponding PCBs. The correlations between measurements with PEDs and immunoassay measurement, converted to GC-based values, are shown in Figure 19.

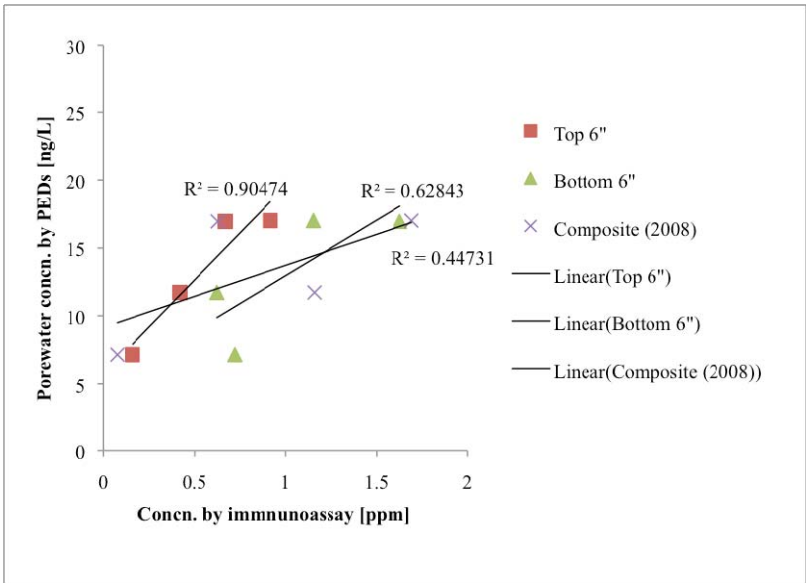

Figure 19. Correlations between converted immunoassay measurements and PED measurements.

Only top 6-inch samples show good correlation with PED-measured pore water concentrations. A better correlation of these samples might be explained by the fact that the PEDs were deployed in the field with the depth of six inches. The immunoassay uses a non-quantitative extraction technique that leads to lower recoveries than conventional sediment extraction and PED measurements. In conclusion, at best, the immunoassay technique is semi-quantitative.

1.3 Quickly and effectively measure pore water PCB concentration in the field

Field Time Series with PE. The PCB uptake curves for PEDs deployed for 239 days from January to September 2007 in Plot C and Plot D show that no equilibrium between PEDs and pore water was reached during the extended sampling period except

for PCBs with a log K_{ow} < 6.2. However, as the majority of contamination comprises PCBs with higher hydrophobicities, uptake is still in the linear phase during the 239-day sampling period.

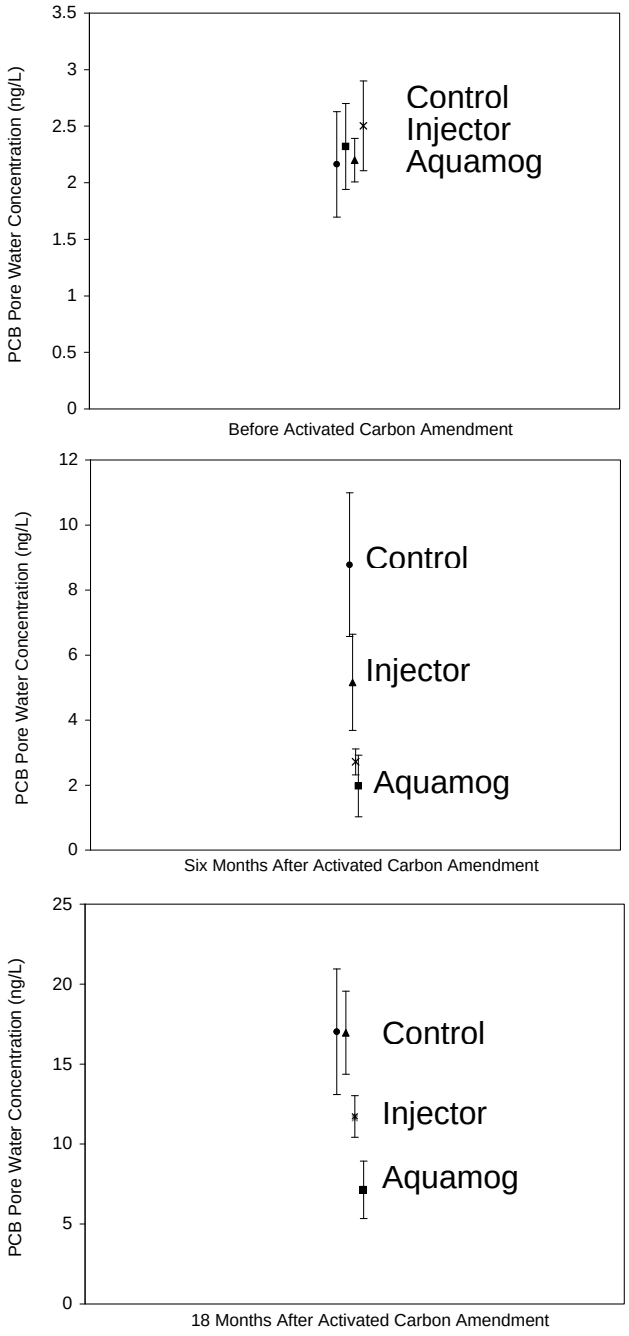

Figure 20. Pore water concentrations estimated using PED uptake and the molar volume adjustment method in four treatment plots before and after activated carbon amendment. Plot C (circles) and Plot E (triangles) are mixed and unmixed controls, and activated carbon was mechanically mixed into Plot D (squares) and Plot F (asterisks) (Tomaszewski and Luthy 2008).

Activated Carbon Amendment Assessment. The molar volume adjustment method was validated by comparison to field measured sampling rates and extracted pore water concentrations. The method was used to estimate pore water concentrations before and after activated carbon amendment of sediment in South Basin at Hunters Point. When the rate of desorption from the sediment is not rate-limiting and uptake kinetics are governed by the aqueous boundary layer (log K_{ow} > 4.4), the amount of HOC in a PED (C_{PED}, ng/kg), is given by

$$C_{PED} = K_{PEW} C_W \left[1 - \exp\left(-R_S t / M_{PE} K_{PEW}\right) \right] \quad \textbf{(16)}$$

where K_{PEW} (L/kg) is the PED-water partitioning coefficient, C_w is the aqueous concentration (ng/L), M_{PE} is the mass of PED (kg) (Booij, Hoedemaker et al. 2003) and R_s is the rate of the PED (L_w/d).

Before amendment, pore water concentrations were similar across the four test plots at Hunters Point. After amendment, a reduction of 60% was noted between average concentrations in Plot C and Plot D. These results show that if mixing does not change the hydrodynamics in the sediment (homogenous structure), large reductions in pore water concentrations can be achieved using AC-amendment. These field tests with PEDs confirm the trend of reduced PCB concentrations in pore water compared to the control plots. Please see Appendix A2 for further explanations of results and figures (Tomaszewski and Luthy 2008).

Vertical pore water profiles. Utilizing equilibrium partitioning coefficients from the literature for POM samplers of 76 µm thickness [(Hawthorne, Miller et al. 2009), K_{POM-76}], vertical pore water profiles in the AC-amended plot (D) at Hunters Point were determined as illustrated in Figure 21a.

Error bars indicate the deviation from the average value since n = 2 for these data points. A sensitivity analysis has also been conducted for the K_{POM-76} values since 17 µm thick POM has been utilized in the present study and it has been shown that K_{POM} values can vary depending on the thickness of the POM. The range of pore water concentrations as measured using POM and assuming equilibrium are also shown in Figure 3a with dashed lines.

Pore water concentrations can also be deduced based on C_{ps} and K_{ps} as well as knowledge of the depletion of the reference compounds and the theoretical basis for these uptake models are described elsewhere (Huckins, Petty et al. 2006; Booij, Vrana et al. 2007). Tomaszewski and Luthy (2008) as well as Harman et al. (Harman, Tollefsen et al. 2008) have also recently presented the theory and modeling as applied to field data for SPMDs and PEDs as described above by equations 2 to 5.

Figure 21b shows the vertical pore water concentration profile for the PE passive samplers. Concentrations are calculated using the K_{PE} values (Tomaszewski and Luthy 2008). Error bars indicate the deviation from the average value since n=2 for these data points. Tomaszewski and Luthy (2008) utilized a molar volume adjustment to account for differences between the PRC and the HOC of interest as presented above. No molar adjustment has been applied here since five PRCs were utilized in this study cover most of the congeners present (94 %) in the field contaminated sediments.

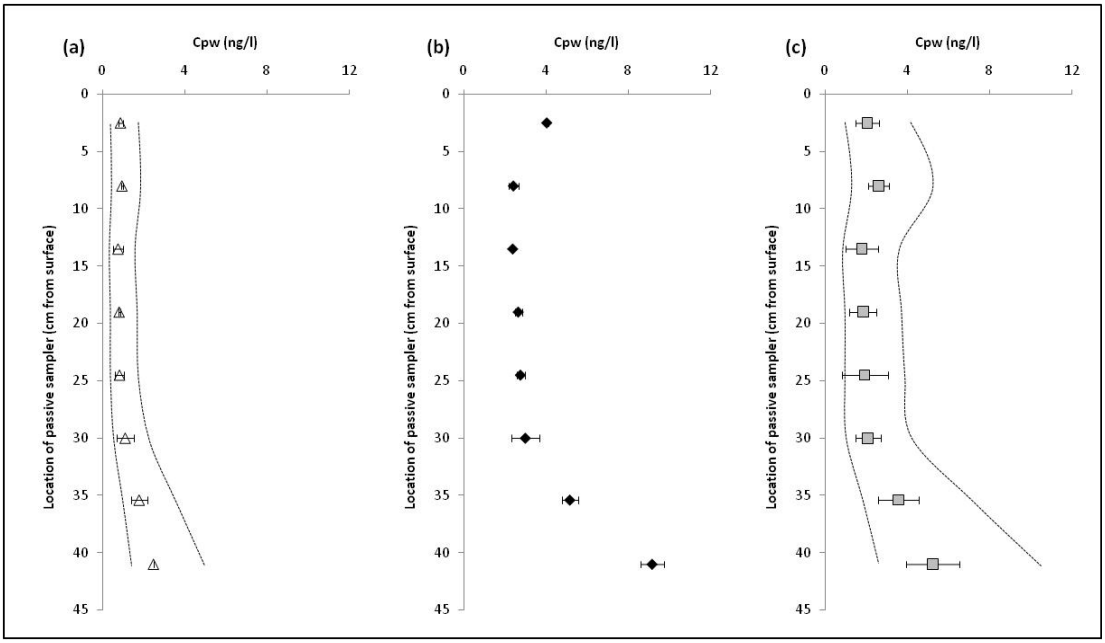

Figure 21. Vertical pore water concentration profiles calculated based on (a) assuming equilibrium has been achieved in the POM after 100 days of exposure, (b) depletion of PRCs from the PE and assuming a first-order process uptake model and (c) depletion of the PRCs from the POM assuming a first-order process uptake model. The dotted lines in (a) and (c) represent the range of concentrations for differences in K_{POM} values due to differences in the thickness of the POM.

Applying these uptake models to SPMDs and PEDs is well established. However, there is less experience with their application and validation for POM since to our current knowledge only one other study has utilized PRCs with POM. Cornelissen et al. (Cornelissen, Okkenhaug et al. 2009) utilized PRCs with POM in order to measure PAHs and PCBs in ground water wells. The authors found > 90% dissipation of the PRCs and therefore assumed equilibrium conditions for calculated the aqueous concentrations. However, in the present study after 5 months of exposure, only 20 – 80 % depletion of PRCs from the POM is observed depending on the degree of chlorination. Therefore the uptake model presented above is also applied to the POM results and illustrated in Figure 21c. The vertical pore water concentration profile represents the average concentrations and standard deviation for the five retrieval data sets.

Figure 22 shows that average PCB concentrations in the top 30 cm of sediment increased from about 0.4 to 2 ng/mg POM and from 0.8 to 1.4 ng/mg PE over the 5 month retrieval time series. Concentrations of PCBs placed deeper than 30 cm ranged from about 1 to 5.6 ng/mg POM and 1.2 to 3 ng/mg PE. These results reflect the distribution of AC in the sediment, where percentages of total organic carbon (TOC) ranged from 3 to 6 % in the top 30 cm (amended with AC) and from 1 to 1.5 % in the deeper sediment (untreated).

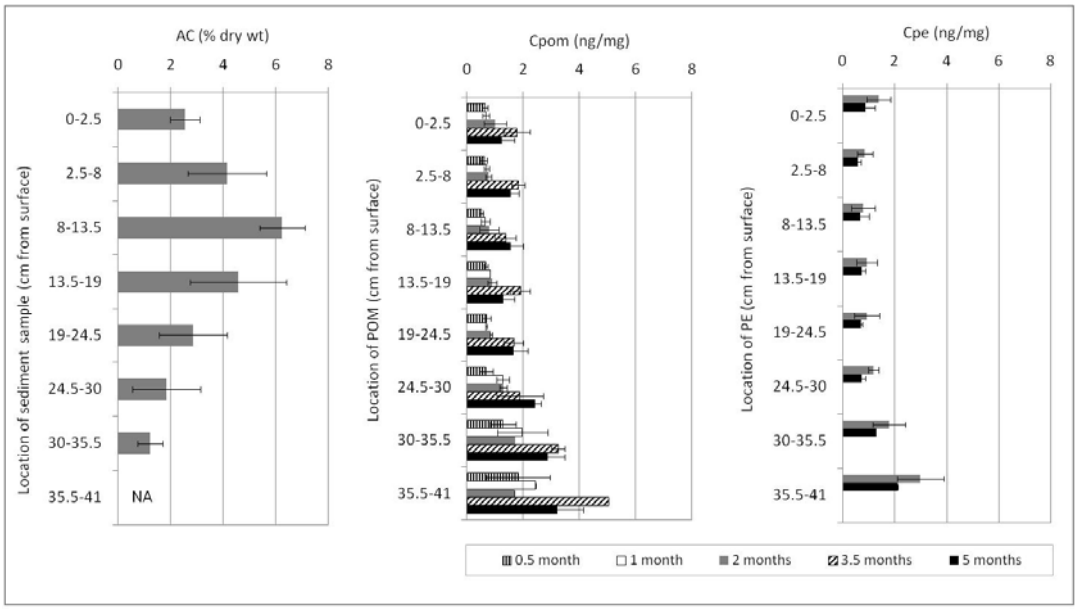

Figure 22. Estimated amount of activated carbon (% dry wt) and measured concentration of total PCBs in the POM and PE samplers deployed into the sediment surface using specially designed sampling rods after five months of exposure in the field.

The differences in PCB concentrations in the POM and the PE after 5 months of deployment also reflect the degree to which the two different passive samplers approach equilibrium. As illustrated in Figure 23, impregnated PRCs show 20 to 80 % depletion from the POM and 2 to 90 % depletion from the PE, depending on the degree of chloronation of the PRC.

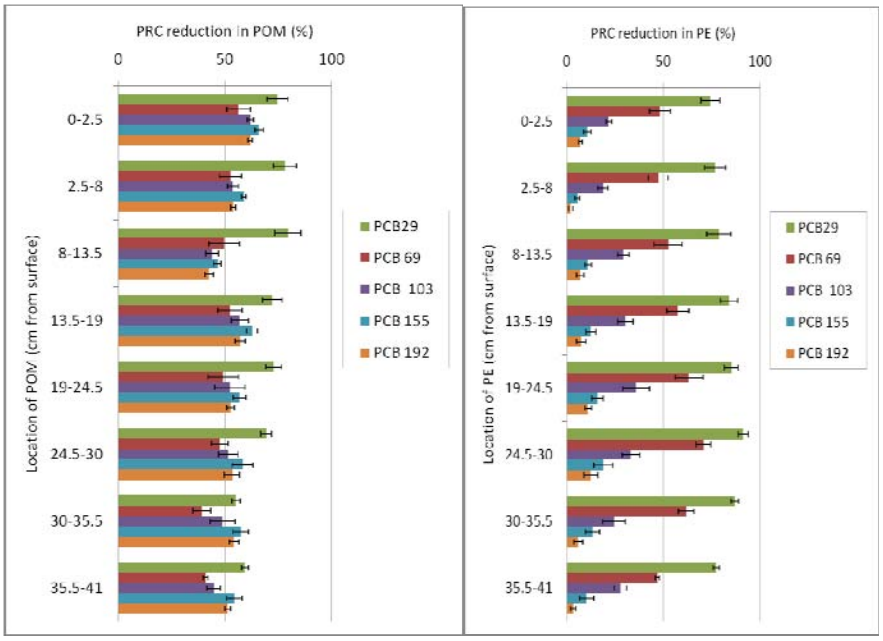

Figure 23. Percent reduction of PRCs as measured in POM (left) and PE (right) after 5 months of deployment.

A manuscript about the work of measuring vertical pore water with the title "Measurement of in-situ PCB pore water concentration profiles in AC-amended sediments using passive samplers" is currently in preparation (Oen, Janssen et al.).

1.4 Demonstrate that PCB pore water concentrations are indicators of mass transfer to activated carbon particles

The pore water measurements and the bioassays suggest that sequestration of PCBs to AC occurs to different extents in the field versus the laboratory. Well mixed laboratory experiment (POM, aqueous equilibrium) show high reductions, indicating that more than 94% of the PCBs are associated with the AC after 28 days mixing. Laboratory bioassays also indicate a reduction in tissue concentration of 73 to 95% suggesting that the PCBs became less available once associated with the AC. Field deployments show less reduction in pore water or bioaccumulation, indicating that in-situ mass transfer is slower or limited due to shorter mixing or influences from the surrounding untreated sediments. When POM sampler or organisms were exposure to AC-amended sediment in small test cages rather than larger test plots in the field, the influence of surrounding contaminated was reflected for both measurements by lower reduction of PCB uptake.

From this and related work, it was concluded that measures of reduced pore water concentration in combination with reduced bioavailability are good indicators for assessing the extent of mass transfer of PCBs to AC particles. Laboratory studies under well-mixed conditions are preferred to assess the potential effects of carbon dose and particle size on treatment effectiveness. In-situ pore water measurements are important to assess the extent of sequestration for the respective field deployment method (mixing regime etc.). Further, bioassays are valuable to reflect remaining availability of PCBs from AC particles, which varies among species depending on feeding strategy.

Table 6. Percent reduction in passive sampling and bioassays from untreated to AC amended Hunters Point sediment.

Method	**Percent reduction upon AC-amendment**
Passive Sampler	
PE (in-situ, 34m^2 test plot)	60
POM (in-situ, worm cages)	37
POM (well mixed, laboratory)	94
Aqueous Equilibrium	99.5
Bioassay	
Clam (freshwater, laboratory)	95
Clam (marine, laboratory)	73
Clam (marine, in-situ, 34m^2 test plot)	50
Oligochaete (freshwater, laboratory)	82
Polychaete (marine, laboratory)	95
Polychaete (marine, in-situ, worm cage))	45

Total PCB concentrations in pore water decreased similarly with increasing AC dose as concentrations in clam tissue. As shown in Figure 24, a highly pronounced dose–effect is observed for aqueous equilibrium PCB concentration and clam body burden.

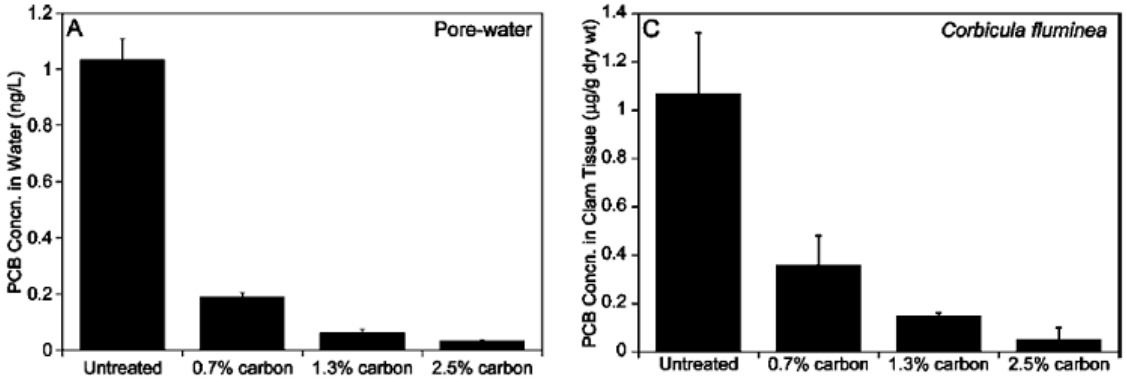

Figure 24. PCB concentrations in water (left) and clam tissue (right) after exposure to untreated sediment and sediment amended with different doses of activated carbon (McLeod, Luoma et al. 2008).

2. Conduct regional surveys to determine the benthic species recruitment pool for Hunters Point

2.1 Define recruitment pool of benthic species, along with model and data needs

Among the traditional ways of determining community structure, the Shannon-Wiener species diversity indices for the reference sites were highest in spring (2.89 ± 0.14), lower in fall (2.09 ± 0.35), and lowest at the fall intertidal reference sites (1.84±0.60). The index was also lower at the reference sites in fall than spring. In both spring and fall, the reference site communities show similar dominant species. The most dominant species was the amphipod *Ampelisca abdita* (median abundance of subsamples >60% of the total abundance of subsamples), which was present at the majority of sites. Unlike the benthic community at Hunters Point, Hunters Point, the second through tenth most abundant species at the reference sites have similarly high abundances (each >1% of the total abundance, Figure 25).

The Shannon-Wiener species diversity index was low at Hunters Point (1.84 ± 0.05), similar to the intertidal reference sites. The benthic community at Hunters Point is dominated by four species of Corophidae amphipods (median abundance of subsamples >60% of the average total abundance of subsamples) and nematodes (median abundance of subsamples >30% of the average total abundance of subsamples). The highest organism abundances were seen at Hunters Point (average of >1,000 nematodes/0.05 m^2). An isopod (*Paranthura japonica*) and a polychaete (*Exogone lourei*) were the only other species with notable abundances (each >1% of the total abundance) and all remaining species were much less abundant (Figure 25).

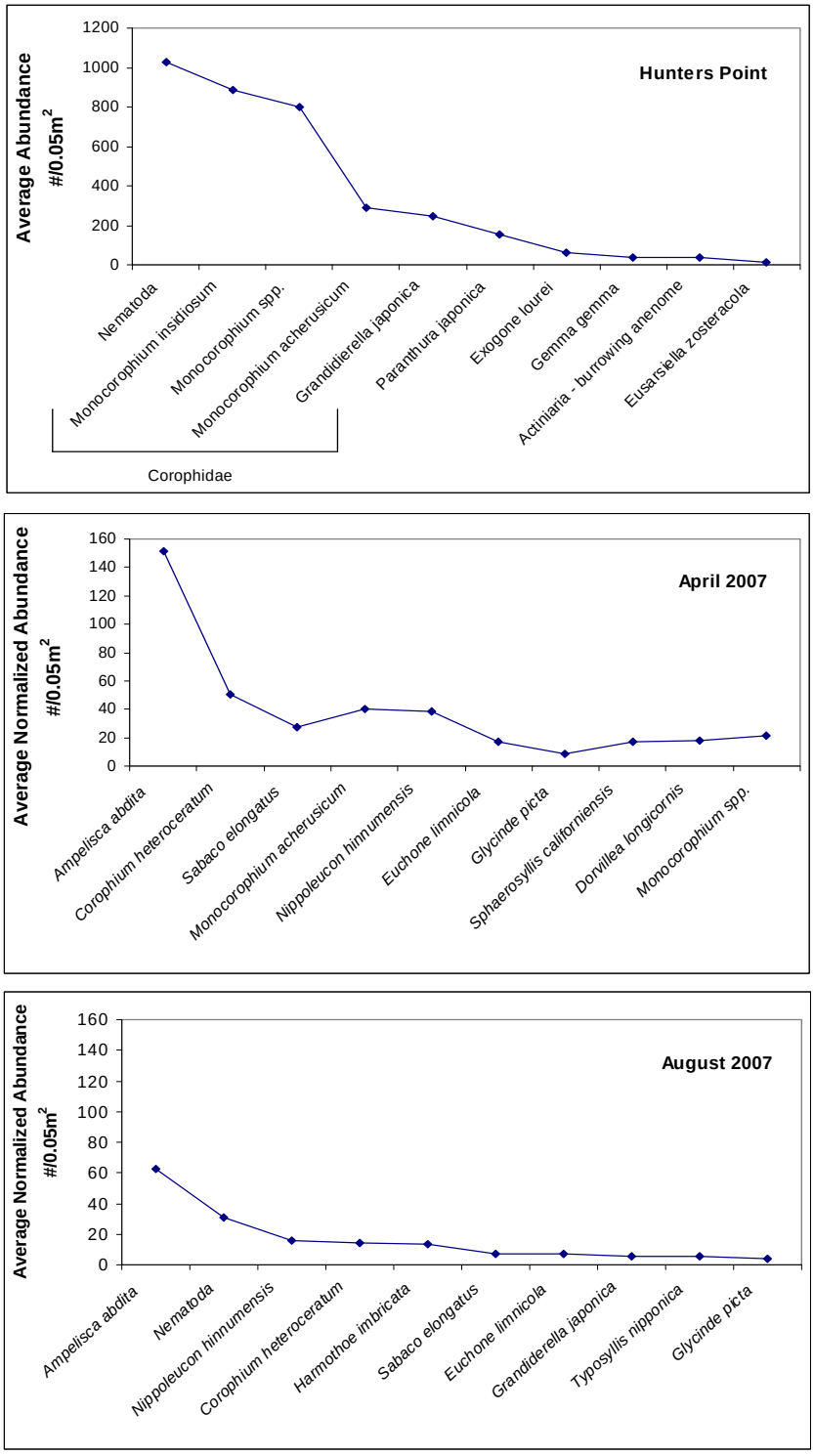

Figure 25. Rank abundance of top ten species at Hunters Point and reference sites. Note that Hunters Point organism abundance is an order of magnitude higher than observed at the reference sites.

Data in Figure 26 show the number of species as a function of total abundance for all individual sample locations and the regional biomes. The benthic communities in the

regional biomes were compared but showed no significant difference. About 80% of all sites, including Hunters Point, had 20 to 30 species (a few had more; a few had less); thus the number of species was not exceptionally different among reference sites, between seasons, or between reference sites and Hunters Point (for all reference stations: spring median of 27 species, fall median of 22 species). The number of species at the intertidal reference sites in fall (median of 21) was very similar to observations at Hunters Point in fall (average of 22 species).

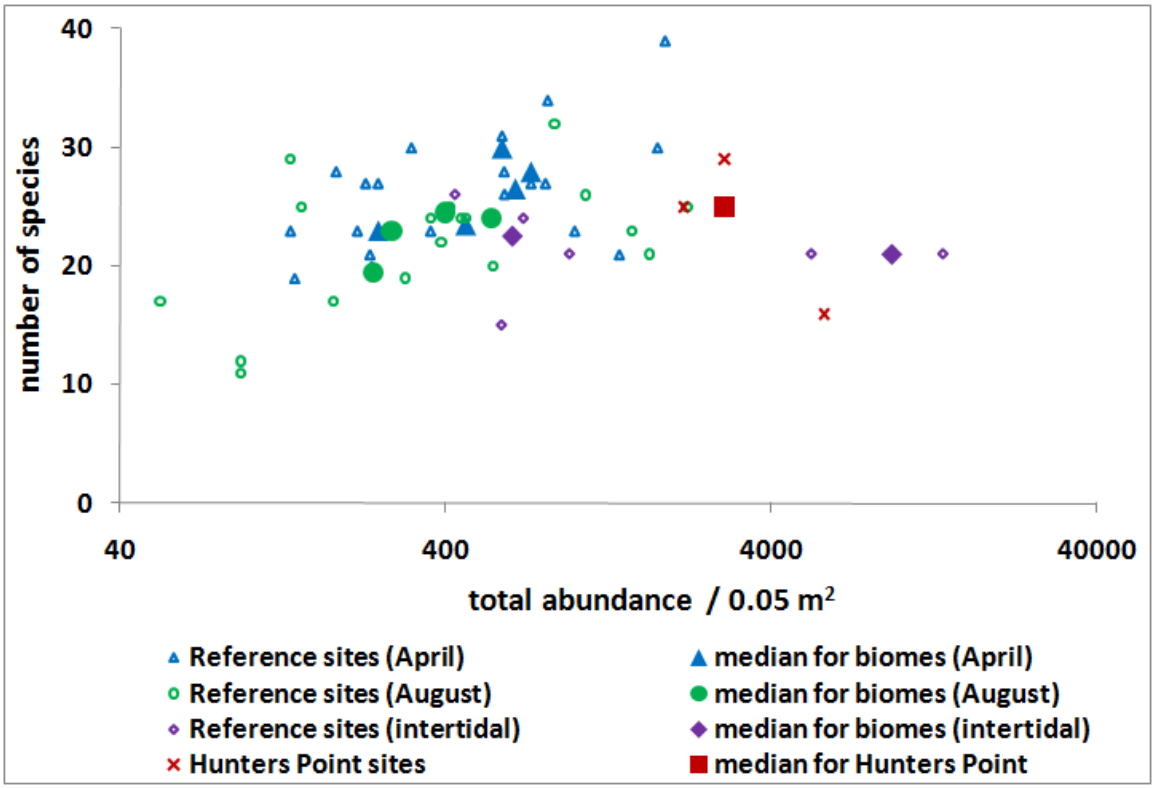

Figure 26. Number of species relative to the total abundance for individual stations (open symbols) and the median abundance of each biome (full symbols) for the reference site assessed in April and August, the intertidal sites, and Hunters Point.

The results of non-metric multi-dimensional scaling, however, shows that the communities at the reference site differ between seasons (April and August) and that the community at Hunters Point differs from reference communities in both seasons. The three data sets in Figure 27 are clustered separately with controls separating by month and both control samples separating from the Hunters Point samples. The stress value (0.04) shows the representation to be excellent as shown in two dimensions.

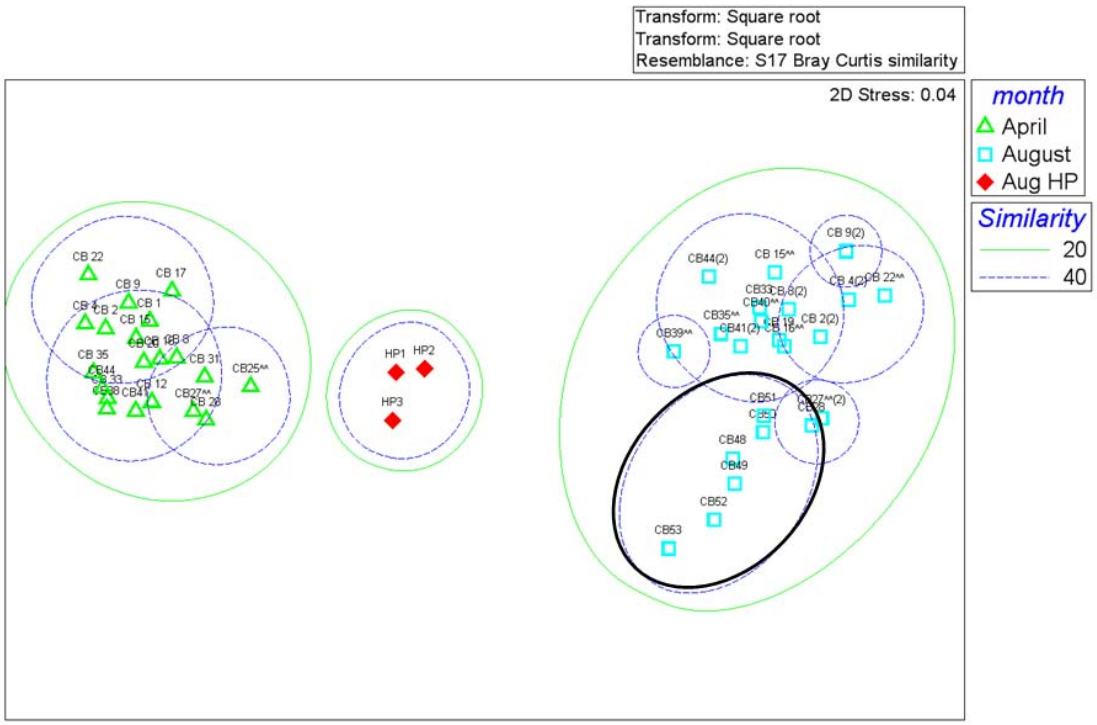

Figure 27. Non-metric multi-dimensional scaling (MDS) plot of the data using Bray Curtis Similarity for total abundance in benthic samples at the reference sites (April and August) and at Hunters Point (August Hunters Point). The black oval surrounds the intertidal sites that can be compared to the Hunters Point samples in red.

A detailed table of species abundance of the sampling site is available at http://www.werc.usgs.gov/BenthicAtlas.

Functional ecology.

Given the similar number of species and diversity indices between sites, it might be concluded that the benthic communities did not differ greatly. While the MDS analysis shows that the community at Hunters Point differs from the reference communities, this representation cannot assess precisely which species or species groups are affected. The functional traits of the species at these sites reveal significant differences between the reference sites and Hunters Point. Data in Figure 28 show the abundance of selected functional groups normalized by total community abundance for the three reference communities (reference April, reference August, reference intertidal) and the Hunters Point community (data table provided in Appendix B). Representative functional groups were selected that show the clearest pathway to contaminant exposure based on their direct interaction with the contaminated sediments, i.e., deposit feeders, subsurface carnivores, egg laying species, and species with no protective barrier. For example, within the feeding groups, the conditions at Hunters Point are most challenging for species that consume sediment (deposit feeders) and their predators (subsurface carnivores).

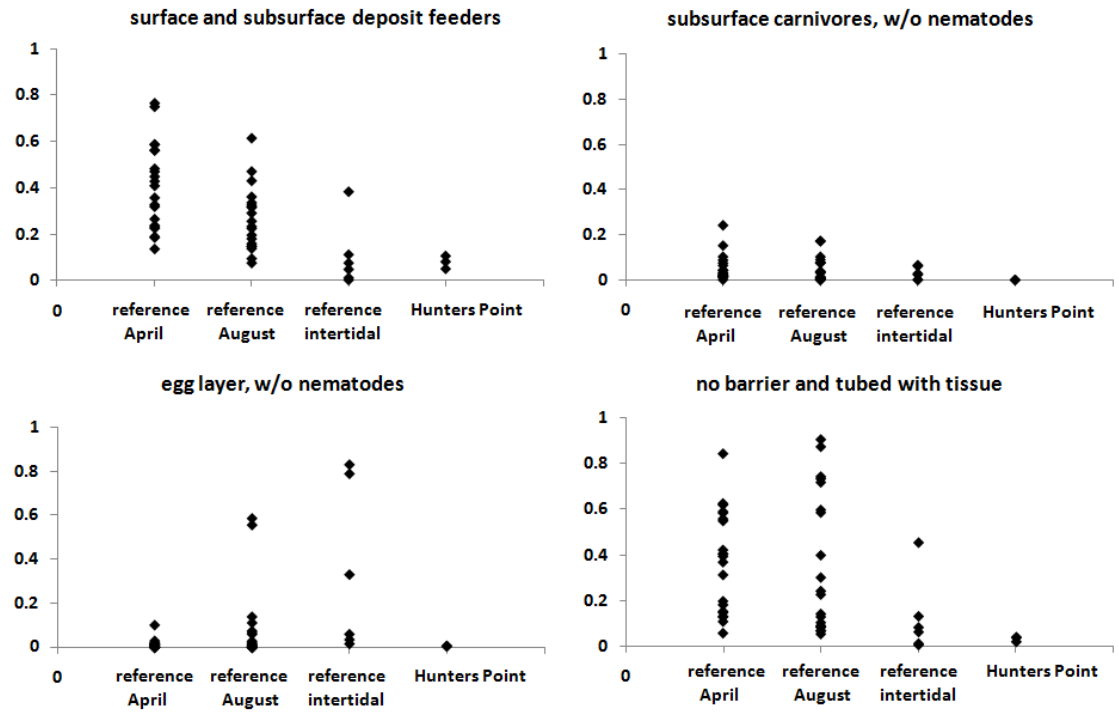

Figure 28. Normalized abundance of species and species groups as the ratio of individual abundance over total abundance for each sample station at the reference sites in April (N = 21), August (N = 20), and at the intertidal sites (N = 6) compared to Hunters Point (N = 3).

The relatively low abundance (≤11%) of deposit feeders at Hunters Point relative to reference sites (≤77%) supports the hypothesis that species that consume the sediment and live with some portion of their body within the sediment are less abundant and differentially absent from Hunters Point (Figure 28A). The percent abundance of deposit feeders at the three reference sites/seasons was not significantly different from each other but different from the percent abundance at Hunters Point (ANOVA, $p<0.10$).

The relative abundance of carnivorous species was similar at the reference sites but the reference sites were dissimilar from Hunters Point (ANOVA, $p<0.05$, Figure 28B). The abundance of subsurface carnivores (overall) was low, probably due to the trophic transfer efficiency typical of such communities (~10%). Thus, also the number of carnivore species was examined and shows significantly less carnivores at HP (1 species) than at the reference sites (3 to 4 species at each site, ANOVA, $p<0.05$). Subsurface carnivores may be less common at Hunters Point either because these species prey on organism with high PCB exposure or due to the lesser abundance of prey.

Species that lay their eggs in the sediment from which juveniles "crawl away" were rather uncommon at all sites, but none were present at Hunters Point (Figure 28C). The lack of a significant difference is likely due to the low overall abundance in this group. Finally, species without a protective barrier or with a weak barrier (tubed with tissue) are significantly less common at Hunters Point (ANOVA, $p<0.05$), with less than 5% of the total abundance, compared to a median of approximately 40% at the reference sites (Figure 28D).

The relative abundance of the functional groups can be compared among six intertidal and the remaining reference stations and Hunters Point. For example, for August 2007 the following patterns can be identified: Surface feeders, either those that filter feed from the water column or deposit feed on the surface of the mud, dominate the feeding groups for all stations. However the benthic community at Hunters Point was entirely composed of this group (100% of the individuals were surface feeders) whereas only 82% of the benthos at the intertidal reference sites were surface feeders and 18% were subsurface feeders. The majority of the individuals at Hunters Point are ovoviparous or viviparous with larvae being brooded and protected before being released as fully viable juveniles on the sediment surface. This fraction is much higher at Hunters Point (97%) than at the intertidal (77%) and remaining reference areas (64%). Hunters Point completely lacks (0%) individuals that lay their eggs on the sediment surface where they develop while in contact with the sediment. This reproductive group is represented by 11% of the individuals in the intertidal and 6% in the remaining reference areas. Finally animals with no barrier between their tissue and the sediment are rare at Hunters Point (3%) while this group is quite common at the intertidal reference sites (46%) and in the remaining reference areas (34%). In general, although some differences between the intertidal and deeper reference areas were observed, much larger differences between the intertidal reference sites and Hunters Point were identified (also compare Figure 27).

Combining the functional traits discussed above, the species that are expected to be found in the reference sites but to a lesser extend at Hunters Point are subsurface/surface deposit feeders or carnivores that lay eggs, which develop into juveniles in the sediment, and which have tissue contact with the sediment. Species with these characteristics are mostly polychaetes and oligochaetes and are found in lower abundances at Hunters Point compared to the reference sites. Seven to eight species with these attributes are present in the reference communities with three species in the intertidal reference communities, including nemertea, nematoda, annelida (oligochaetes and mainly polychaetes), and mollusca. In comparison, only one species with all these attributes was present at Hunters Point (a polychaete) and it was represented by only two individuals.

Details for the total and relative abundances of the other functional groups can be found in the Appendix B. A technical manuscript will be prepared by Janet Thompson et al. titled “Using functional ecology to examine pollution effects on benthic communities: examples with a long term study and a spatially intensive study” (Thompson, Luoma et al.). A discussion of the chemical and physical properties of the sediment at Hunters Point and at the reference sites can be found under task 4.1.

3. Biodynamic modeling to predict PCB uptake by benthic organism

3.1 Conduct laboratory experiments to define physiological coefficients for biodynamic model

Bioaccumulation was measured and the biodynamic model was parameterized and tested for the marine clam *Macoma balthica*, the freshwater clam *Corbicula*

fluminea, and marine polychaete *N. arenaceodentata*. Table 7 summarizes the physiological parameters.

Table 7. Physiological parameters for benthic invertebrates and biodynamic modeling.

Parameter, symbol unit	*Neanthes arenaceodentata*	*Macoma balthica*	*Corbicula fluminea*
Filtration rate, FR L water/ g dry wt / d		2[c]	45[c]
Aqueous assimilation efficiency, AE_{aq} %		50[c]	20[c]
Aqueous uptake rate constant, k_w L water / g dry wt / d, = FR x AE_{aq}	0.5[a]	1	9
Ingestion rate, IR g sediment / g dry wt / d	3.5[b]	0.25[c]	0.03[c]
Sediment assimilation efficiency, AE_s %	7 [b]	20[c]	20[c]
Elimination rate constant, k_e 1 / d	0.04[a]	0.05[c]	0.04[c]
Growth rate constant, k_g 1 / d	0[b]	-	-

[a] (Janssen, Croteau et al. 2010). [b] (Janssen, Oen et al. 2010). [c] (McLeod, Luoma et al. 2008).

Facultative deposit-feeding clams. The results for the measurement and modeling of PCB bioaccumulation from Hunters Point sediment for a freshwater and marine clam have been published by McLeod et al. (2007 and 2008). The results demonstrate that adding AC to sediment from Hunters Point can reduce PCB uptake by these clams by 67 to 95%. The biodynamic model could predict bioaccumulation in the clams as presented in Figure 29.

The model can also be used to investigate the relative importance of dietary and aqueous uptake routes for PCB accumulation by the two species. As shown in the data in Figure 30, the relative contributions of these two uptake routes to PCB body burden differ markedly between *M. balthica* and *C. fluminea* in our test systems with untreated sediment. For *C. fluminea*, aqueous uptake is almost equally important to dietary uptake. However, *M. balthica* appear to take up approximately 90% of their PCB body burden through sediment ingestion. Similar patterns are seen when modeling uptake from treated sediment (not shown). These findings strongly suggest that PCB reductions in the aqueous phase alone are not sufficient to effectively reduce PCB uptake by sediment-dwelling organisms.

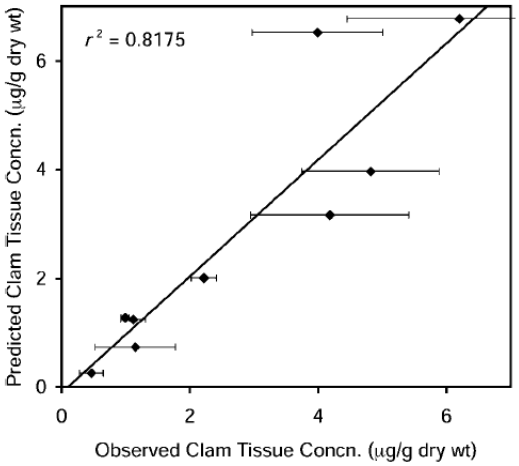

Figure 29. Predicted clam tissue polychlorinated biphenyl concentrations using the biodynamic model versus experimental observations. Error bars represent one standard deviation (*n* = 3–5). Solid line represents the linear regression between predicted and observed concentrations (McLeod et al. 2007).

Additionally, these findings have implications for the choice of monitoring species for sediment quality assessment. As demonstrated in Figure 30, deposit-feeders such as *M. balthica* may be more reflective of the local sediment environment than filter feeders such as *C. fluminea*.

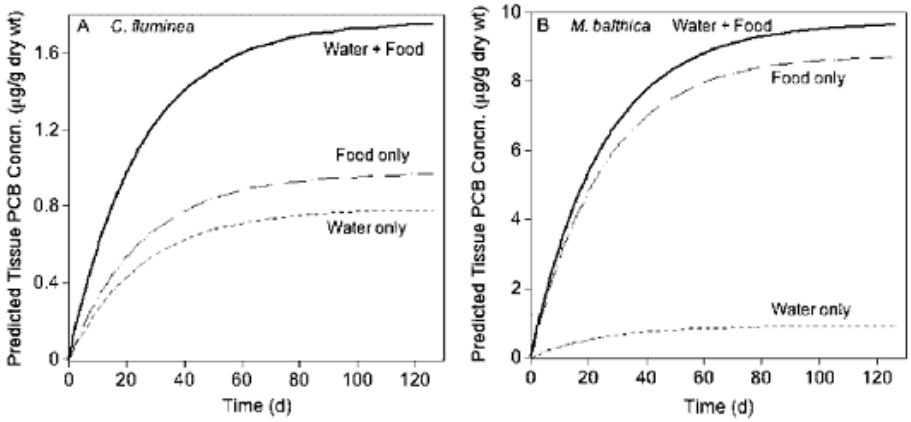

Figure 30. Relative contributions of PCB uptake via food and water for *M. balthica* in untreated Hunters Point sediment as predicted with the biodynamic model (McLeod et al. 2007 and 2008).

Especially in a flow-through riverine system, organisms like *Corbicula* will integrate more regional sediment and water quality through their filtering of water that originated upstream of the test site. Therefore, if the goal of a monitoring study is to assess local sediment conditions in open systems, it is strongly recommended to use sediment-ingesting benthic organisms like *M. balthica* or *N. arenaceodentata* as discuss in the following.

Deposit-feeding polychaetes. Bioaccumulation was measured and the biodynamic model was parameterized and tested for the marine polychaete *N. arenaceodentata*. The results have been published by Janssen et al. (2010). PCB tissue concentration increased exponentially during the 28-day exposure to untreated sediment and exceeded the sediment concentrations after 14 days (Figure 31, left). Tissue concentrations were reduced by 95% by the AC-amendment of the sediment (Figure 31, right). Despite the absence of net increase in PCB tissue concentration through exposure to AC-amended sediment relative to the initial body burden (t = 0), the change of homolog distribution in the organism's tissue suggests exchange of background PCBs (tetra-homolog dominated) for sediment-associated PCBs (hexa- and hepta-homologs dominated). Thus, there was minimum, residual bioavailability of higher chlorinated PCB congeners.

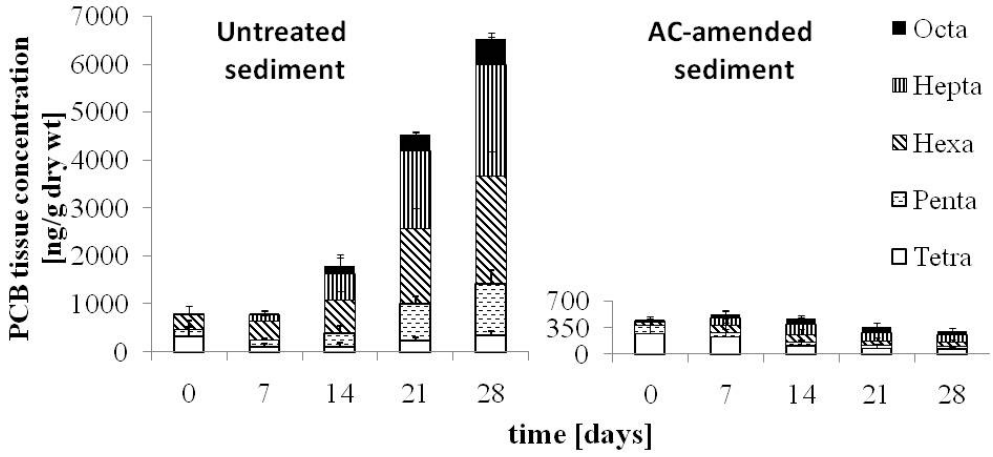

Figure 31. PCB tissue concentrations of *N. arenaceodentata* for contaminants accumulated from Hunters Point site sediment (left) and AC-amended Hunters Point sediment (right). Error bars represent one standard deviation, showing five homolog groups of prominent PCBs (Janssen et al. 2010); (Data table in Appendix B, 6).

No significant difference between weights in AC-amendment (15.5 ± 5.2 mg, N = 39) versus untreated Hunters Point sediment (14.5 ± 12.4 mg, N = 26) was observed at day 28 ($p < 0.05$). There was no significant temporal trend for lipid content of *N. arenaceodentata* (standardized by wet weight) during the sediment microcosm experiments. However, the lipid content of organisms grown in the AC-amendment was on average three times less compared to for organisms from untreated sediment ($p < 0.05$). These effects cannot be explained with our study. Perhaps natural organic matter (NOM) in sediment becomes less available after AC addition.

The biodynamic model was parameterized for all coefficients but the assimilation efficiency (AE) from sediment and AC-amendment. The assimilation efficiency was estimated solving the biodynamic model for AE with the measured time-series bioaccumulation. Figure 32 shows the time-dependency of the AE from sediment which could be explained by an increase in gut residence time (GRT) with the growth of the worms. Longer GRTs translate to more complete sediment extraction. Penry and Jumars classified polychaetes by their gut architecture, and simple organisms with just one gut

compartment can be modeled as a plug flow reactor (Penry and Jumars 1990). The GRT increases during growth because the gut volume increases faster relative to the mass ingested, which is a consequence of the cylindrical shape and organism's length increasing at a rate faster than organism's diameter (Janssen, Croteau et al. 2010).

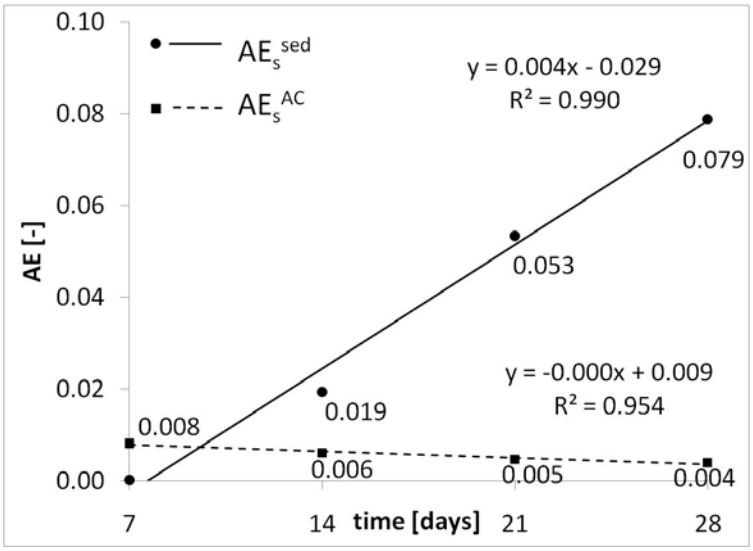

Figure 32. Assimilation efficiencies (AE) from sediment (AE_s^{sed}) and activated carbon (AE_s^{AC}) calculated for *N. arenaceodentata* with an average ingestion rate of 8.7 g/g dw per day (Janssen et al. 2010).

With the increasing AE values, the predictions of the biodynamic model were in good agreement with the measurements and the model also captures the effect of reduced PCB bioavailability after AC-amendment (Figure 33). The biodynamic model wherein diet was the primary source of bioaccumulation explained PCB accumulation in these polychaetes during the bioassays. Although changes in pore water concentration after AC-amendment of the sediment often correlate with reduced PCB bioaccumulation, the reduced availability of contaminants from ingestion of sediments appears to be the actual cause of lower tissue concentration.

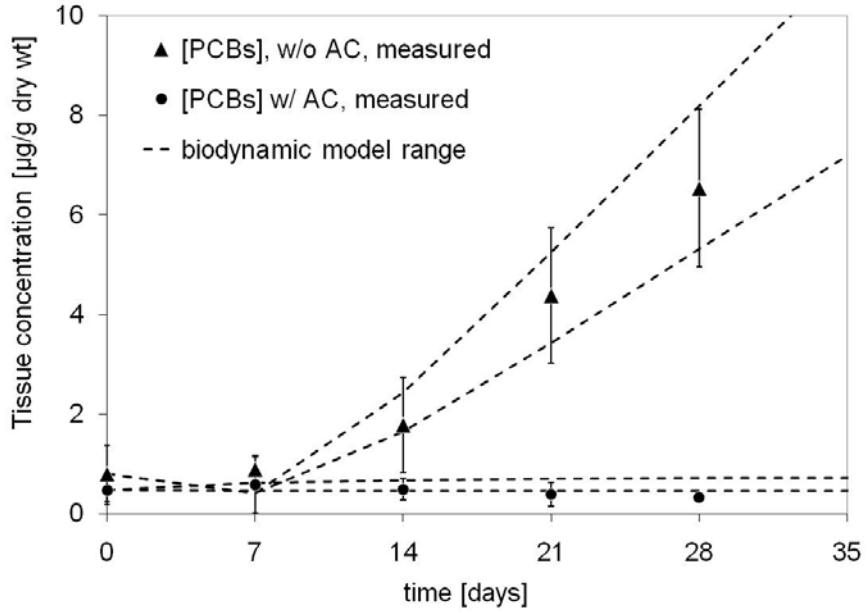

Figure 33. Measured values and modeled range of PCB tissue concentrations for *N. arenaceodentata* in sediment w/ and w/o AC-amendment considering the upper and lower bounds for sediment IR (Janssen et al. 2010).

Direct uptake from ingested sediment appeared to account for more than 97% of the total PCB accumulation, partly because uptake from solution at environmental concentrations was very slow compared to uptake from food. The deposit-feeding polychaete appears to be a useful test organism to evaluate PCB exposure from sediment and reduced contaminant availability by remediation such as the AC in-situ amendment.

3.2 Demonstrate applicability of biodynamic model in the field

PCB uptake by biota and passive sampler. Under laboratory conditions, high PCB bioaccumulation of 3.2 ± 0.43 µg/g dry wt was observed within 14 days exposure to untreated Hunters Point sediment. The data in Figure 34A show that PCB bioaccumulation with caged polychaetes was about half as large under field conditions compared to laboratory conditions with 1.8 ± 0.22 µg/g dry wt and 1.7 ± 0.42 µg/g dry wt for in-situ deployments at Hunters Point in February and July, respectively with no statistically significant differences between the two field tests (t-test, $p > 0.05$). Polychaete bioaccumulation of PCBs was significantly reduced with AC-amendment under field deployment conditions to 0.9 ± 0.15 µg/g and 1.2 ± 0.45 µg/g for in-situ tests in February and July, respectively, and reduced further under laboratory conditions to 0.3 ± 0.04 µg/g for the ex-situ tests (t-test, $p < 0.05$, Figure 34B).

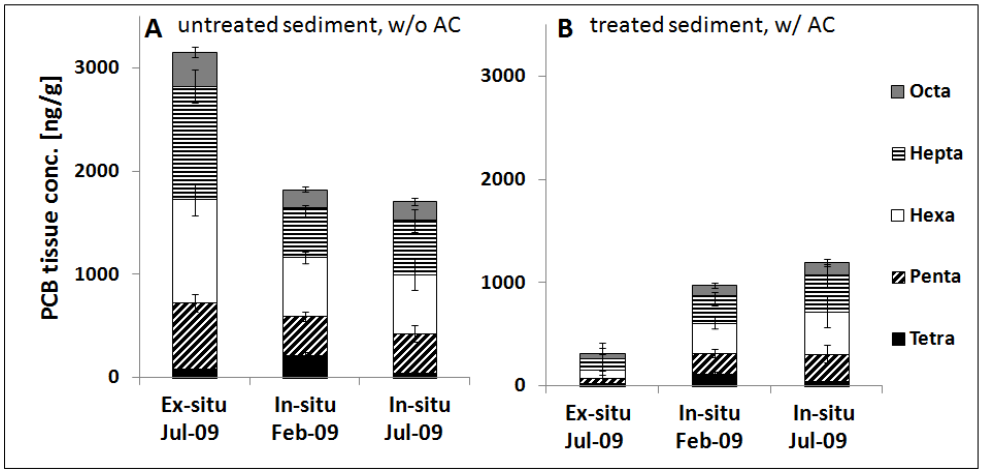

Figure 34. PCB tissue concentrations on dry wt basis after 14-day in-situ and ex-situ bioassays with untreated Hunters Point sediment (A) and AC-amended sediment (B) showing the predominant homolog groups (on average > 99% of total PCBs). Error bars represent one standard deviation, N=4-5 composites of 10-12 organisms each; (data table in Appendix B, 7).

The uptake of PCBs into POM samplers positioned in the subsurface sediment (3 cm) was significantly reduced by AC-amendment for both field tests (t-test, $p < 0.05$, Figure 35). The PCB uptake into POM from untreated sediment was similar at the surface layer (0.5 cm) and subsurface (t-test, $p > 0.05$). However, with AC-amended sediment POM samplers placed in the surface layer showed significantly greater uptake compared to POM samplers placed in the subsurface (t-test, $p < 0.05$).

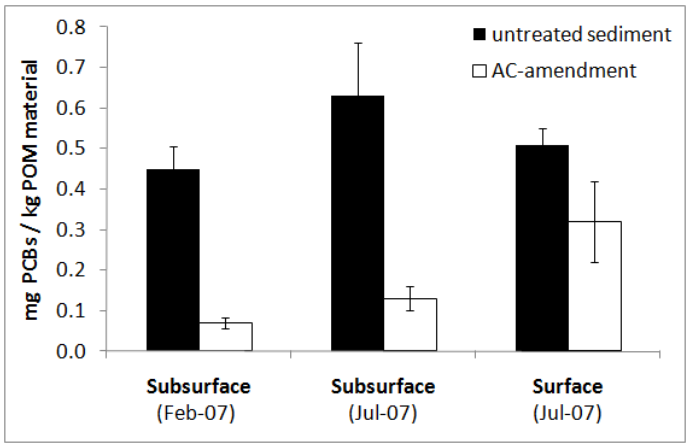

Figure 35. In-situ PCB uptake to POM samplers in deployed cages with untreated and AC-amended sediment, with POM samplers placed in the subsurface sediment (3 cm) and surface sediment (0.5 cm); (data table in Appendix B, 8)

The relative reduction of impregnated performance reference compounds (PRCs) in the POM placed in the subsurface ranged from 7 to 27% for the untreated sediment and from 8 to 35% for the AC-amended sediment depending on the hydrophobicity (K_{ow}) of the PRCs. Comparisons between the individual PRCs for untreated and AC-treated

sediment show no significant differences between the percent reductions (t-test, $p > 0.05$). Similarity in depletion of PRCs indicates a similar water sampling rate for these passive samplers, which allows a comparison to PCB uptake even though the POM samplers are not at equilibrium.

Reduction of uptake by biota and passive sampler after AC-amendment. The data in Figure 36A show the relative reduction of bioaccumulation by AC-amendment for the predominant PCB homolog groups. The reduction in bioaccumulation after AC-amendment was greater under laboratory conditions with about 90% reduction compared to the in-situ tests where only 40 to 48% reduction was observed.

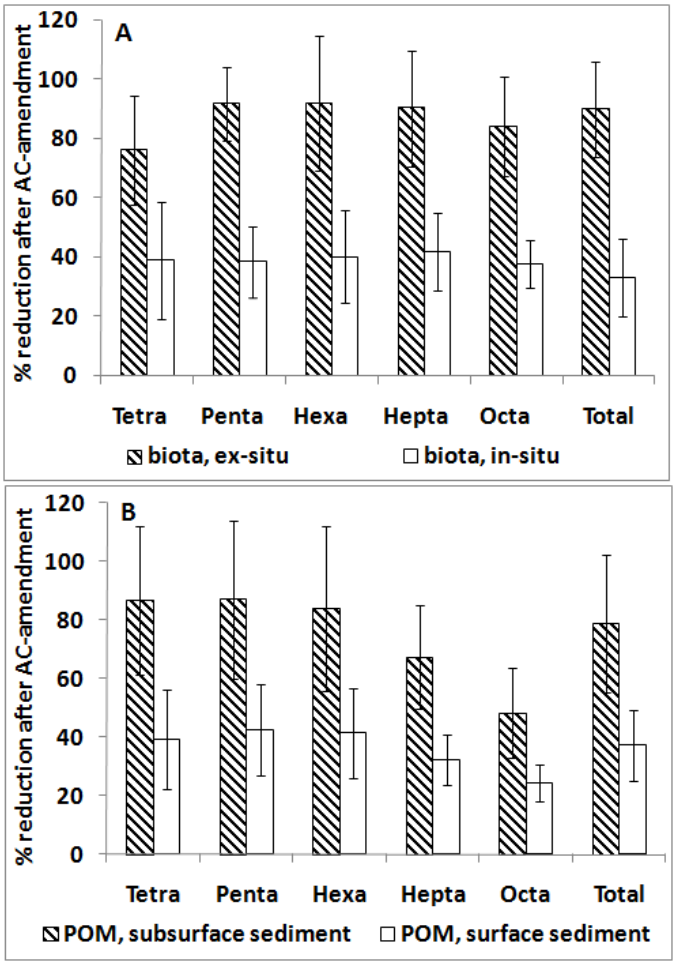

Figure 36. (A) Relative reduction in PCB uptake after AC-amendment for polychaetes (*N. arenaceodentata*) for ex-situ and in-situ bioassays and (B) Relative reduction in PCB uptake in POM samplers deployed on sediment surface and in the subsurface. Error bars represent one standard deviation.

Uptake into POM samplers deployed in the subsurface sediment was reduced by almost 80% in the AC-amended sediment in the field (Figure 36B), while POM samplers placed in the surface layer showed only 37% reduction of uptake. In summary, the relative reduction of uptake in surface POM measurements correlate with in-situ

bioaccumulation reduction, and subsurface POM measurements correlate with ex-situ bioaccumulation reduction.

Model estimates for PCB exposure and feeding. The assimilation efficiency (AE) from sediment in the laboratory tests was estimated as 7% for untreated sediment and 0.7% for AC-amended sediment, and these relative values agree with previous laboratory results (Janssen, Croteau et al. 2010). The in-situ ingestion rates, IR, were not measured in this study. Assuming similar AE from untreated sediment under laboratory and field conditions, the in-situ IR can be estimated as 3 to 4 gram sediment per gram dry weight per day, which is less than half of the ex-situ IR (8.7 g/g per day). The apparent in-situ AE of PCBs was estimated as 2 to 3%. A mass balance approach was used to estimate the fraction of ingested AC-amended sediment relative to overall ingestion of sediment. With $AE_{AC\text{-}amendment}$ being 0.7% and $AE_{untreated\ sediment}$ being 7%, the organism fed about 55 to 70% of the time on the untreated sediment as it would occur with feeding on incoming sediment deposits in the field cages.

Bioaccumulation from untreated sediment for in-situ and ex-situ exposures. The PCB uptake by polychaetes from untreated sediment was about 45% less in the field tests compared to laboratory tests. Bulk deposit-feeders like *N. arenaceodentata* accumulate more than 95% of PCBs from ingested sediment and only a minor fraction is assimilated directly by transport across the tissue from the aqueous phase (Janssen, Croteau et al. 2010). A sensitivity analysis of the biodynamic model shows that bioaccumulation is most sensitive to changes of sediment concentrations (C_s), sediment organic carbon fraction (f_{oc}), assimilation efficiency from sediment (AE), and ingestion rate (IR) followed by growth rate (k_g) (See Janssen et al., 2010, Supporting Information, Table S3).

The major difference between the in-situ and ex-situ bioassays is the dynamic exchange with the surrounding water and surficial material at the field site. Due to wave-motion and tidal cycles, water in the field-deployed cages was continuously exchanged with the overlying water and fine particulates from the surrounding field site. Almost 50% of the Hunters Point sediment comprises silt or clay (< 62.5 µm, (Battelle 2004)), which is able to pass through the screen openings of the cages (105 µm). The process of surrounding surficial material entering the cages is important because it can modify exposure conditions and food availability.

In this study, POM samplers were deployed in the top 0.5 cm, which represents the layer influenced by surficial material. The passive sampler measurements revealed similar PCB uptake from the surface and subsurface of untreated sediment (Figure 3). It can be inferred that the PCB availability from surficial material and untreated sediment are similar and thus the values determined for the biodynamic model (C_s and AE) were not greatly affected by incoming material. These observations are supported by a previous analysis of the surficial material for the Hunters Point site ((Cho, Ghosh et al. 2009), Supporting Information, Figures S5 to S8). Cho et al. reported that PCB concentration, TOC content, and back carbon (BC) content in the surficial material (top 0.5 cm) is within the range of the underlying contaminated sediment (1 to 2 µg PCBs / g dry wt, 0.7 to 1.5% TOC by dry wt, and about 0.003 g BC/g sediment).

Under similar exposure conditions from underlying and incoming sediment, bioaccumulation from untreated sediment in the field must have been affected by other processes, e.g., the organism's activity like feeding and growth behavior. Significantly different growth rates between organisms in the laboratory and field were measured and have important implications for growth dilution, which decreases tissue concentrations. Despite the absence of organism's growth in the field, lower in-situ tissue concentrations were observed.

In addition to contaminant availability (C_s, AE) and growth (k_g), bioaccumulation is most sensitive to sediment ingestion rate. Bulk deposit feeders like *N. arenaceodentata* compensate for lower food quality with greater sediment IR (Cammen 1980; Cammen 1989). The surficial material in the field is most likely a combination of sediment with low food availability and particulate organic matter with a high food availability (e.g., plant or animal decay, microbial biomass, phytoplankton, or microalgae) (Cammen 1989). The assumption of higher food availability in the field is supported by previous analysis of the surficial material at the Hunters Point site that showed a difference of C-13 isotope signature between the surface and subsurface sediment (Cho, Ghosh et al. 2009). These observation suggests differences in origin and/or biological age between the materials even though no increase of organic carbon (OC) could be observed ((Cho, Ghosh et al. 2009), Supporting Information, Figures S5 and S6).

As field measures of IR were not practical with our study design, the biodynamic model was used to estimate the in-situ IR as 3 to 4 g/g per day, which is about 50% less compared to ex-situ conditions (8.7 g/g day). A 1% increase in food availability, for example, would lower the IR to these estimates (see details in Supporting Information). Thus, it is most likely that differences between laboratory and field tissue concentrations are predominantly related to reduced in-situ feeding.

Bioaccumulation from AC-amended sediment for in-situ and ex-situ exposures.

AC-amendment reduced polychaete bioaccumulation significantly in all tests but to a greater extent under laboratory conditions compared to field conditions (90% vs. 40 to 48%, respectively). Field tests with POM samplers and AC-treated sediment showed greater extent of PCB uptake into POM samplers positioned in surface sediment compared to samplers positioned in the underlying sediment. The uptake to POM samplers in the surface sediment was similar in field tests for all cages, whether treated sediment or untreated sediment. Thus, the higher PCB availability in the surface layer reinforces the hypothesis that surficial material re-deposited in the cages posed a similar PCB exposure as the untreated sediment.

The reduction of uptake by POM samplers deployed in the surface sediment layer correlated with the in-situ bioassays. On the other hand, reduction of uptake by POM samplers deployed in the subsurface correlated with the ex-situ bioassays. These observations indicate that organisms in cages with AC-amended sediment were exposed to a mixed diet of surficial material with higher PCB availability and higher food quality than the underlying AC-amendment with lower PCB availability and lower food quality. In-situ influences by surrounding sediment have been reported previously for caged organisms, for example, in comparison to laboratory tests the field data showed less reduction in PCB bioaccumulation by clams after AC addition because the test organisms

were partially feeding on surface deposits in the field (Besten den, Naber et al. 2003; Cho, Ghosh et al. 2009).

The in-situ bioavailability of PCBs from a mixed exposure to AC-amended sediment and surficial material was estimated with the biodynamic model as 2 to 3%, which was much greater than from AC-amended sediment measured under controlled laboratory conditions (0.7%). It was estimated that the organisms fed 55 to 70% of the time on the surficial material, which seems reasonable for deposit feeders. Surface feeding was observed during the laboratory tests.

Additional influences by environmental parameters not considered in this study (e.g., sun intensity, bioturbation, or stress by water turbulences) have to be acknowledged. Exposure to other pollutants can influence organisms in the field as elevated nickel and mercury contaminations has been reported for South Basin (Battelle 2004). The redox conditions in the field may influence the availability of those metals. Nevertheless, PCBs are known to be the predominant pollutant of concern at the study site.

4. Development and application of a predictive general ecosystem recovery model'

4.1 Predict ecological recovery and compare to field studies

Sediment characteristics. The desired information about the selected sediment characteristics was acquired and matched with the sampling locations (biomes) of the benthic community surveys. The information for the fraction of fines (<0.0625 mm) was obtained from the same reference stations as used for sediment chemical concentrations. The stations and respective TOC content within these reference areas are listed in the Appendix B as provided by the San Francisco Estuary Institute. The sediments have similar texture, being dominated by fines (< 0.062 mm) with less than 20% sand and a TOC content within the range of 1 to 2% of the dry weight. The TOC values are similar for Hunters Point and the reference sites with 1.4 ± 0.4% and 1.1 ± 0.2%, respectively. The fraction of fines is also similar with 80 ± 20% at Hunters Point and 77 ± 11% at the reference sites and all sample locations are within the polyhaline area of the San Francisco Bay. The contaminants considered in this study are present at all sample locations.

A previous study by Chapman et al. used one site in the San Pablo Bay with similar contamination levels as the reference sites here to assess pollution-induced changes in the San Francisco Bay (Chapman, Dexter et al. 1987). The variability of sediment contamination between the six biomes in this study is very low suggesting that the sites experience similar chemical exposures. Thus, the same average sediment concentrations from all 30 reference sites can be further considered as the reference conditions for the HPS site.

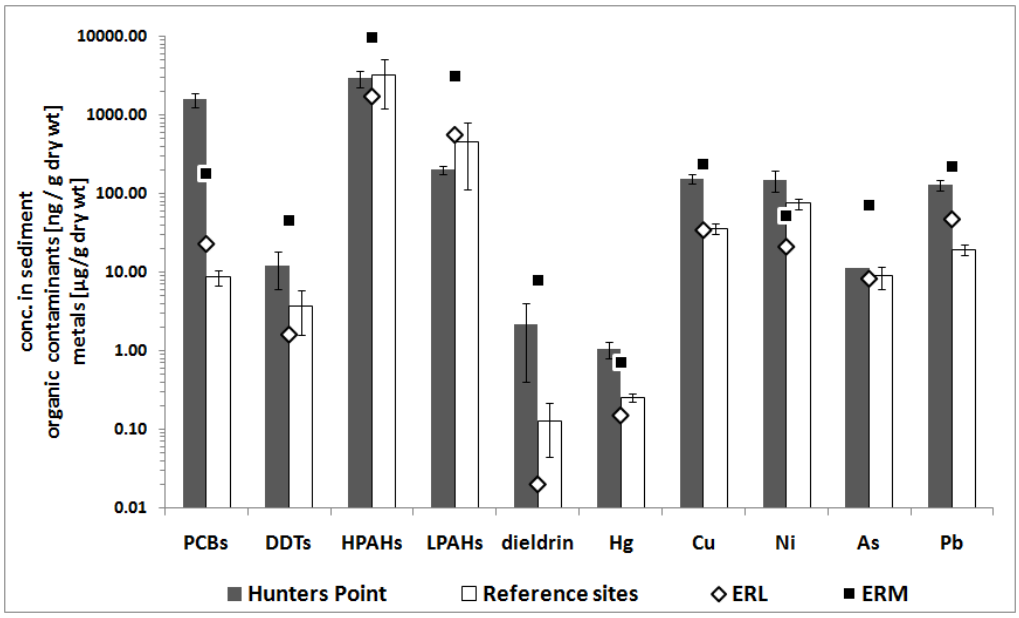

Figure 37. Average sediment concentrations of predominant pollutants at Hunters Point and on average at the reference sites in the Central Bay. The sediment quality guidelines thresholds of contaminant-specific ERL (open diamonds) and ERM (full squares) are shown for organic contaminants [ppb] and metals [ppm]. Error bars represent one standard deviation.

Besides LPAHs, all contaminant concentrations exceeded the ERL threshold at all sites and in addition, PCB and lead are below the ERL value at the reference sites. Nickel concentrations exceed the ERM threshold of 51 ppm at the HPS site and for the reference conditions. In addition, PCBs and mercury exceed the ERM values of 180 ppb and 0.71 ppm, respectively. The concentrations for PCBs are most distinct between the reference conditions and the HPS site. At the HPS site, total PCBs reach 1570 ± 325 ppb and even higher concentrations of approximately 7000 ppb were reported further north-east at South Basin (Battelle 2004). The average, total PCB concentration at the reference sites is 8.8 ± 2 ppb about 180 times lower or 0.6% of the concentration at HPS.

The ERMq values, expressed as the ratio of measured sediment concentrations divided by the ERM value, for the HPS site show that mainly PCBs exceed the SQG by a factor 8.7, followed by nickel (factor of 2.9) and mercury (factor of 1.5). For the reference conditions only the ERMq for nickel is elevated (factor 1.5). The ERMq for the ten contaminants considered (as listed in Figure 37) for the HPS site is six times higher compared to the reference conditions (ERMq = 2.5). The relative contribution of each contaminant is presented in Figure 38 and shows for Hunters Point that PCBs contribute 56% to ERMq, followed by nickel (20%), and mercury (10%). PCBs only contribute 2% to the ERMq at the reference sites whereas nickel (57%), mercury (14%), and HPAHs (12%) are the more dominant pollutants (Figure 38).

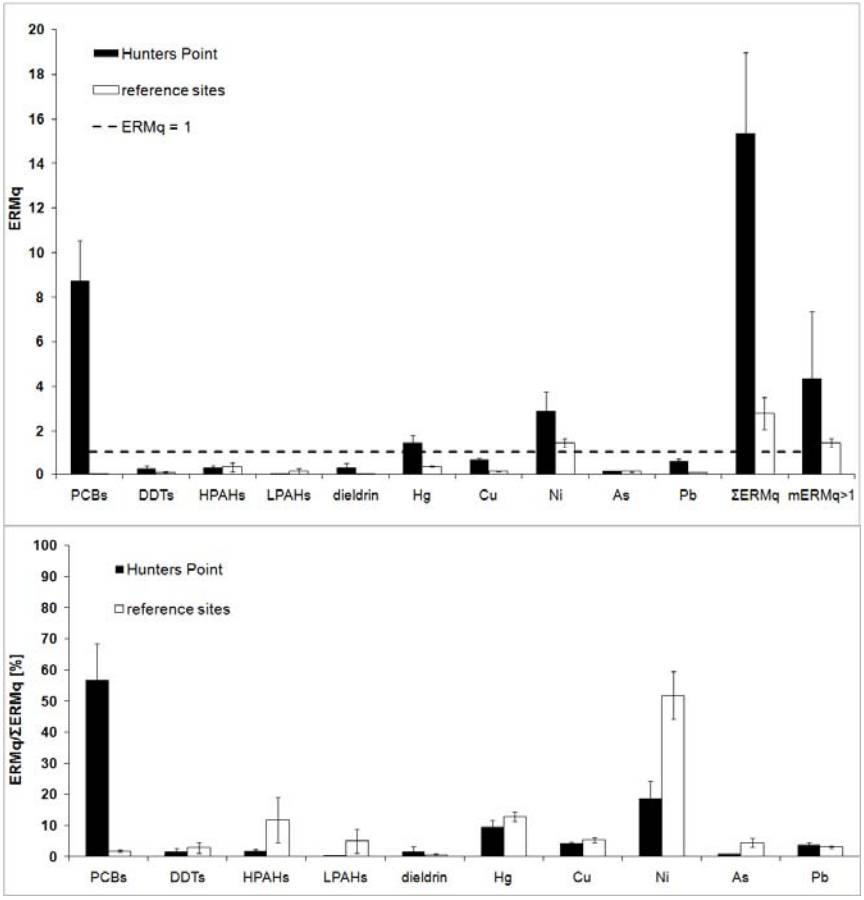

Figure 38. Top: Effective range Medium quotients (ERMq) for predominant sediment contaminants and the sum of ERMq (ERMq) and the mean ERMq considering only contaminants with ERMq >1 for Hunters Point (N = 6) and the reference sites. (N = 30).

Greater cumulative chemical stress at the HPS site is also emphasized by the mERMq>1 of 4.4 compared to 1.5 for the reference conditions. The mERMq considering all contaminants included in this study, are 1.5 for the HPS site and 0.3 for the reference conditions. The Regional Monitoring Program (RMP, data from 1993 to 2005) assessed mERMq values throughout the San Francisco Bay and reported a range from 0.05 to 0.4 (Melwani and Thompson 2007). The sediment information for the station considered in this study is based on the assessment from the RMP and represents the low- to mid-range of mERMq in the San Francisco Bay. Note, that the RMP assessment considered additional contaminants, which were mainly present in low concentrations. As discussed before, including many, less abundant contaminants for calculating the mERMq reduces the overall value, and has to be considered when comparing between different study designs. Previous tests show that a mERMq below 0.5 is related to few incidences of toxicity based on standardized amphipod toxicity tests, while an mERMq of 1.5 suggests that incidence of toxicity are more than twice as frequent (Long, Ingersoll et al. 2006).

In conclusion, the sites selected appear to represent appropriate reference conditions in the Central Bay with minimal pollution. While other pollutants are present at Hunters Point, the predominant chemical stressors are PCBs, which suggests that PCB-induced adverse effects in wildlife are very likely. This conclusion is supported by "The

Validation and Feasibility Studies for Hunters Point" (Battelle 2004; Brajas 2008), which suggested that the risk estimated from copper, mercury, and PCBs were identified as the primary risk drivers to the surf scoter at the HPS site (Brajas 2008). The Validation Study concluded that PCBs are of greatest concern at the site and the present concentrations pose potential risk to birds feeding on benthic invertebrates and fish (Battelle 2004). Even though, the SQG do not allow prediction of the specific extent of ecological risk, the data presented demonstrate that the 30 evaluated reference sites show similar chemical exposure and this exposure is significantly lower compared to the HPS site with PCBs being the risk driver at HPS. Previous studies demonstrated that the use of the ERMq can identify and rank locations with ambient levels of chemical stressors which then provide valuable reference sites for the analysis of potential influences on the benthic community structure (Hyland, Balthis et al. 2003).

Concept of Ecosystem recovery potential. The predictions of the biodynamic model for PCB tissue concentrations under different exposure conditions are presented in Figure 39. The results show a varying exposure and bioaccumulation for three classes of organisms with different functional feeding strategies that reflect the variation of interactions with the contaminated environment. The polychaete *N. arenaceodentata* acquires food by bulk deposit feeding and accumulates about 9000 ng PCB per gram dry weight when exposed to Hunters Point sediment, which is four times more than the facultative deposit feeding clam *M. balthica* and forty times more than the filter-feeding mussel *M. edulis,* which acquires food exclusively from the overlying water.

The model predictions also demonstrate that the PCB tissue concentrations at Hunters Point are an order of magnitude greater than exposure to the guideline values of ERM for PCB sediment concentrations. PCB tissue concentrations expected at the reference sites are two orders of magnitude less than at Hunters Point for all feeding groups. The data in the log-log plot show that the predicted PCB tissue concentrations decrease in accordance with decreasing PCB sediment concentrations because the sediment desorption kinetics are expected to be similar for the sites compared and over this concentration range. The PCB tissue concentrations predicted for exposure conditions according to the ERL threshold of the SQG are only slightly higher than for the reference sites in the Central Bay and about 0.5% of the present body burden at Hunters Point for all three organisms. The tissue concentrations predicted upon AC-amendment are similar to the clean-up goal and the ERM conditions, which is one order of magnitude lower compared to present conditions at Hunters Points.

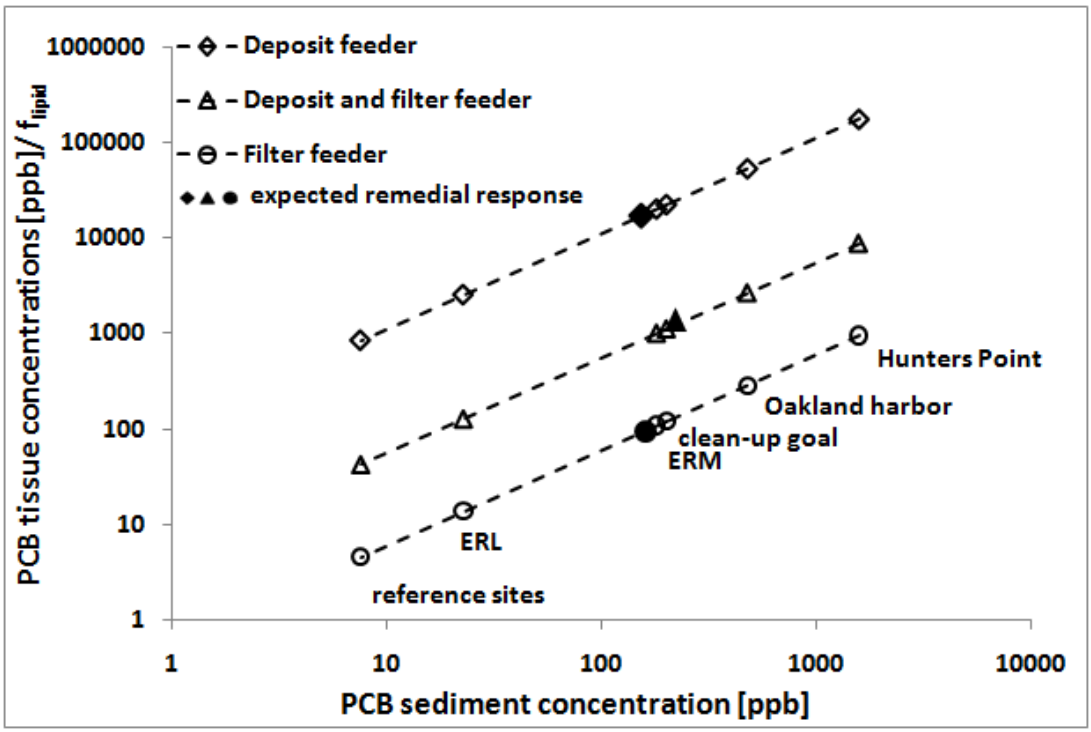

Figure 39. Lipid-normalized PCB tissue concentrations estimated with the biodynamic model for *Neanthes arenaceodentata* (deposit feeder), *Macoma balthica* (surface and deposit feeder), *Mytilus edulis* (filter feeder) under different exposure condition corresponding to the following: Hunters Point, a hot spot at Oakland Harbor, the clean-up goal for Hunters Point, thresholds suggested by the sediment quality guidelines (ERL and ERM), and the reference conditions in the Central Bay.

The biodynamic model was then used to estimate the required reduction of PCB availability at Hunters Point to lower PCB tissue concentrations in the respective species to concentrations under different exposure conditions (Figure 40).

The PCB availability has to be reduced by 60 to 75% to reach concentrations similar to the exposure conditions at the hot spot at Oakland harbor. Ninety percent reduction of the Hunters Point PCB availability would achieve PCB tissue concentrations that would comply with the sediment quality guideline (SQG) ERM threshold and the clean-up goal for Hunters Point. The expected remedial response after AC-amendment predicts that the PCB tissue concentration would remain slightly higher than present at the reference sites. To lower PCB tissue concentrations to levels estimated for the reference sites, the PCB availability has to be reduced to almost zero (>99%), which is not feasible with the tested AC amendment. The expected remedial response upon AC-amendment at Hunters Point would reduce PCB availability by 85 to 90% which corresponds to exposure conditions of the ERM threshold and the defined clean-up goal at Hunters Point (Brajas 2008).

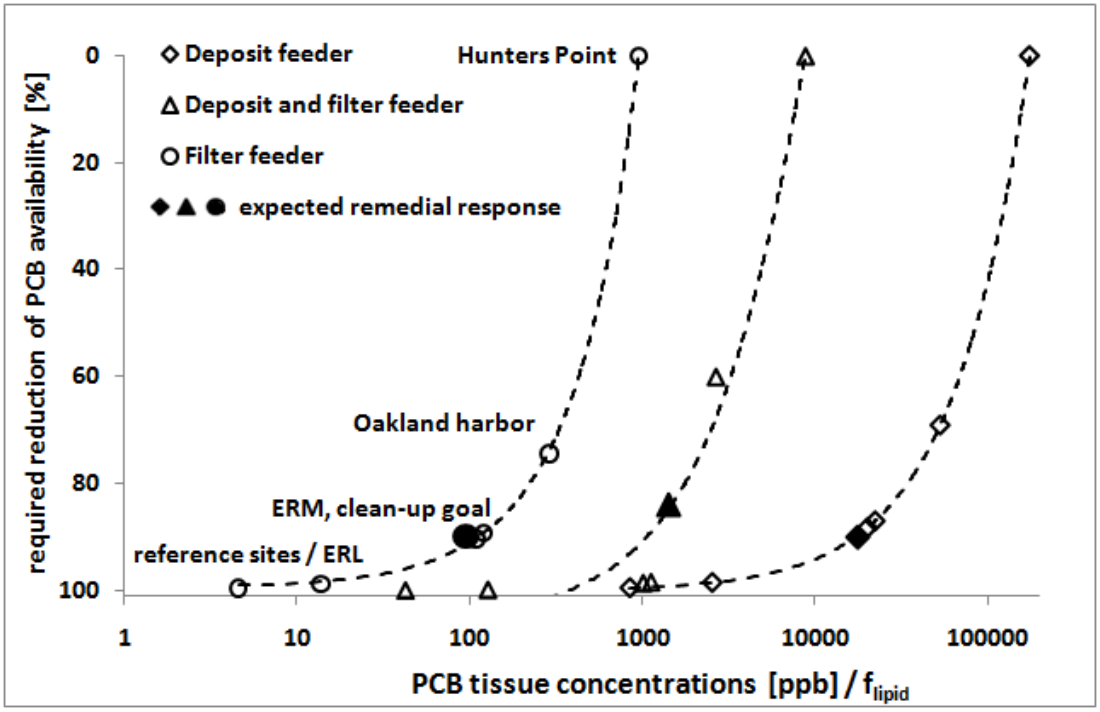

Figure 40. Required reduction of PCB availability at Hunters as estimated with biodynamic model to achieve desired lipid-normalized PCB tissue concentrations for *Neanthes arenaceodentata* (deposit feeder), *Macoma balthica* (surface and deposit feeder), *Mytilus edulis* (filter feeder) under corresponding to exposure conditions at Hunters Point, a hot spot at Oakland Harbor, the clean-up goal for Hunters Point, thresholds suggested by the sediment quality guidelines (ERL and ERM), and the reference conditions in the Central Bay. The expected remedial response after AC-amendment is indicated by full symbols. Dashed-lines lineate the polynomial (2nd order) trend of the data.

Since no information of benthic community data is available for hypothetical exposure conditions that are comparable with the ERM or the clean-up goal, a more detailed estimate of the response of the benthic community composition at Hunters Point upon AC-amendment cannot be made. However, the SQGs are based on observed biological effects after acute and chronic bioassays tests, and are designed to protect wildlife. Hence the predicted remedial response to the AC-amendment suggests that ecosystem recovery is very likely.

4.2 Correlate conventional alternative assessment methods and model ecosystem recovery in the field

The Shannon-Wiener **s**pecies diversity index was highest in spring at the reference sites (2.89 SE 0.14). The species diversity index in fall at the reference site was 2.09 (SE 0.35) and 1.84 at both the fall intertidal reference sites and Hunters Point (reference site SE 0.60 and Hunters Point SE 0.05). The number of species was not exceptionally different between reference dates (spring median for all reference stations of 27 species; fall median of 22 species). The number of species at the intertidal

reference sites in fall (median of 21) was very similar to that observed at Hunters Point in fall (average of 22 species for the subsamples, Figure 26).

On average, the number of species and the range in abundance of species at the Hunters Point and at the reference sites did not greatly differ (Figure 26). Given the similar diversity index between the reference sites and Hunters Point, it may be concluded that the benthic communities did not differ greatly. However, in examining the functional groups of these sites, it was found that there are significant differences between the reference sites and the Hunters Point site.

Diversity indices and the abundance of indicator species were not sensitive enough to recognize the changed community composition at Hunters Point. The analysis on the basis of functional ecology identified the lack of deposit feeders, the absence of egg laying species and lack of species with no barrier or a weak barrier at Hunters Point relative to the 30 reference sites. The comparison of functional ecology resulted in the most meaningful analysis approach to identify changes in the community composition for this study.

5. Monitor and assess carbon amendment in Grasse River

5.1 Bioaccumulation studies with *Lumbriculus variegates*

Laboratory bioassays with the aquatic oligochaete *Lumbriculus variegatus* and Grasse River sediment showed a rapid increase in tissue concentration that equilibrated after about 10 days exposure. The amendment of the sediment with 2.5 % (dry wt) AC could reduce PCB uptake by more than 80% after 28 days (Figure 41).

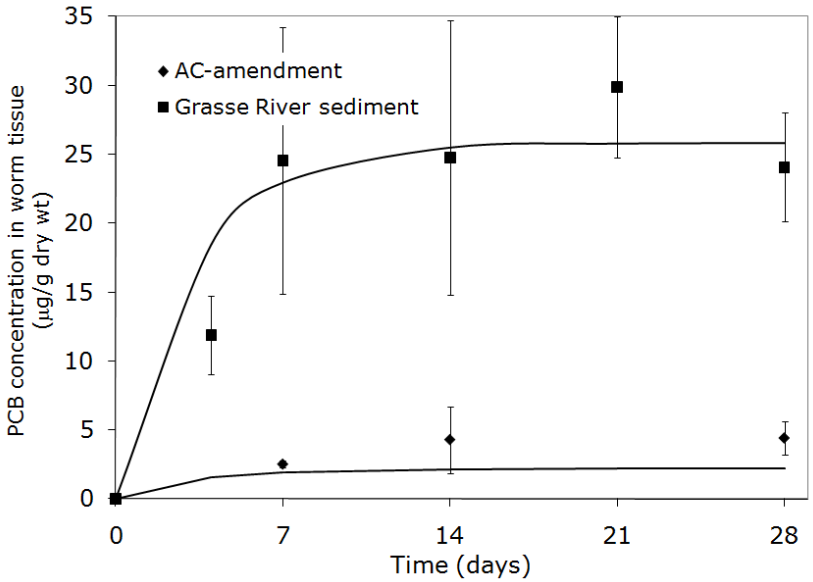

Figure 41. PCB tissue concentrations for *Lumbriculus variegates* in Grasse River sediment and AC-amendment and biodynamic model (-).

The biodynamic model successfully predicted total PCB uptake over a 28-day time series.

5.2 PCB mass transfer modeling with heterogeneous mixing

Temporal variation in the sediment temperature profiles. The data in Figure 42 to Figure 49 show temporal variations of sediment temperature at various depths with solar radiation and tides for 14 days in August 2007 and March 2008. Each temperature data set showed clear diurnal patterns with variations depending on tidal cycle and solar radiation. These diurnal patterns were previously observed in a tidal mudflat in Korea by Cho et al. (2005). At Hunters Point, temperature fluctuations were greater at shallower depths in the mud. Sediment surface temperature (2.5 cm depth) was highest at mid-day, when the solar radiation was highest and lowest just before sunrise. Seasonal differences between the two sampling periods were evident with temperatures at the lowest sediment layer (60 cm) 5°C, higher in August 2007 than in March 2008. This difference was probably due to the higher temperature of overlying seawater during the summer.

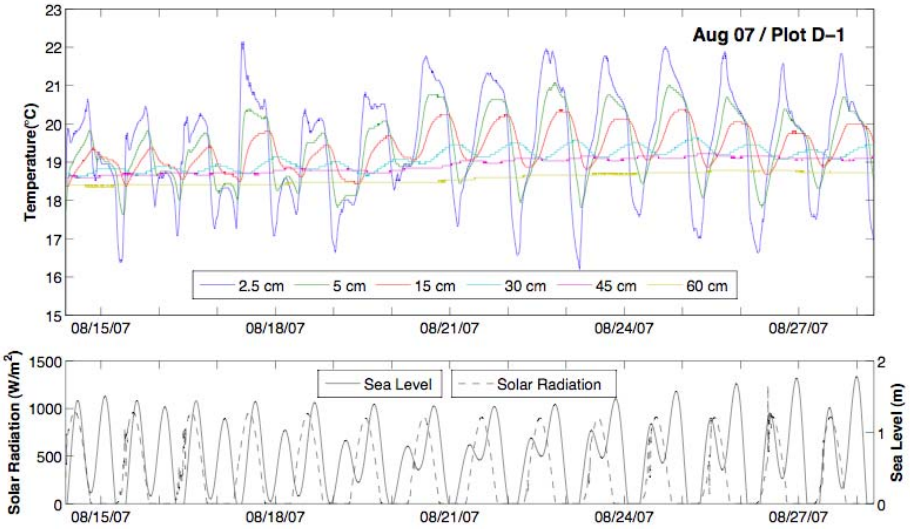

Figure 42. Temporal variations in the sediment temperatures (colored lines) and the sea level (black solid line) at Plot D-1, and the solar radiation (black dotted line) for 14 days in August 2007.

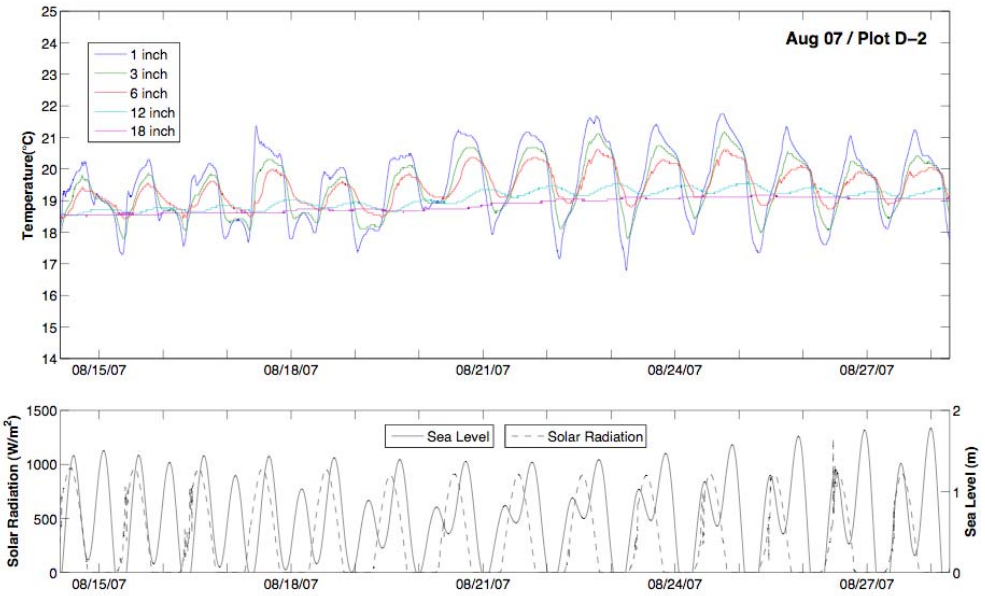

Figure 43. Temporal variations in the sediment temperatures (colored lines) and the sea level (black solid line) at Plot D-2, and the solar radiation (black dotted line) for 14 days in August 2007.

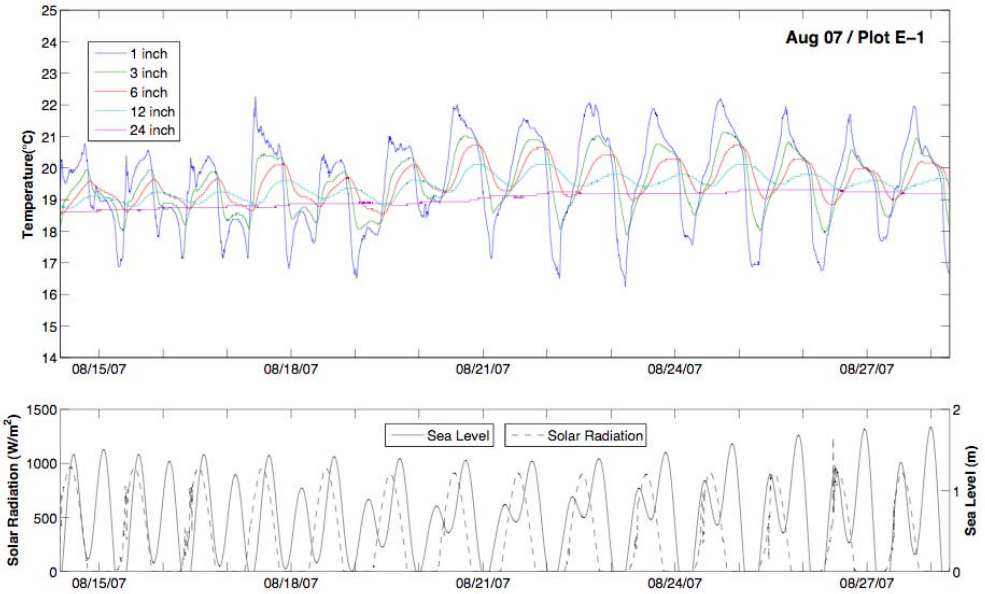

Figure 44. Temporal variations in the sediment temperatures (colored lines) and the sea level (black solid line) at Plot E-1, and the solar radiation (black dotted line) for 14 days in August 2007.

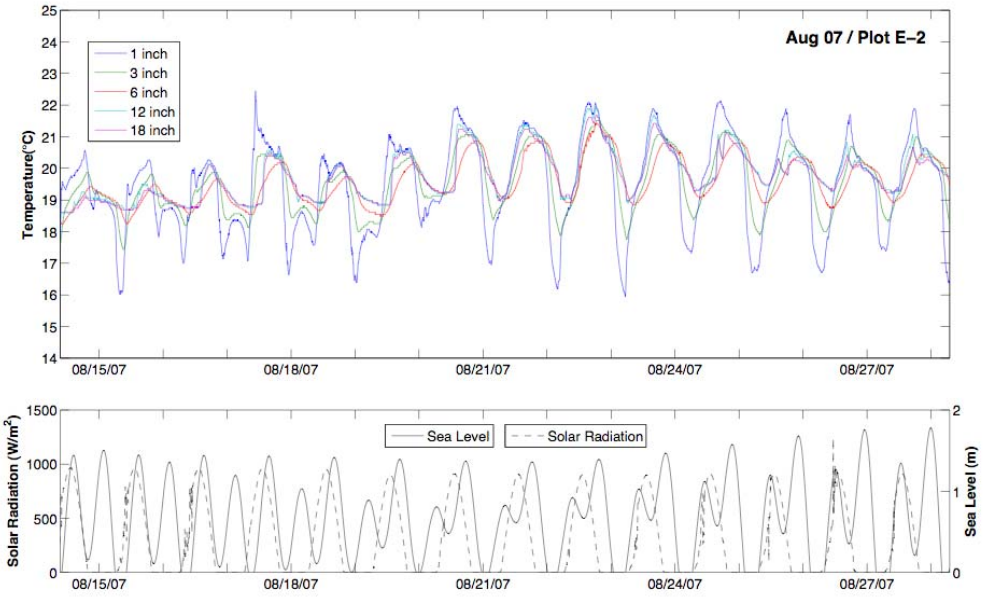

Figure 45. Temporal variations in the sediment temperatures (colored lines) and the sea level (black solid line) at Plot E-2, and the solar radiation (black dotted line) for 14 days in August 2007.

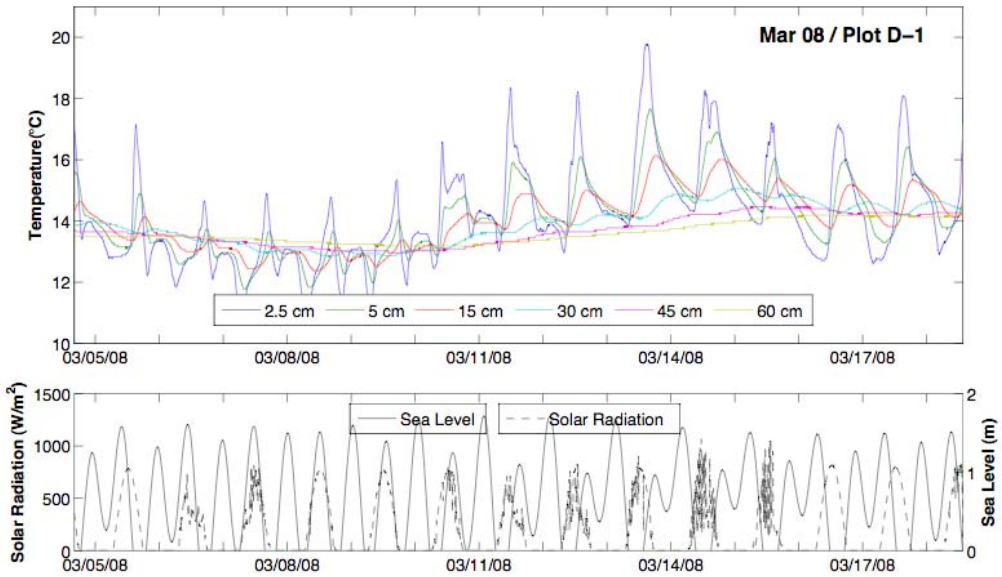

Figure 46. Temporal variations in the sediment temperatures (colored lines) and the sea level (black solid line) at Plot D-1, and the solar radiation (black dotted line) for 14 days in March 2008.

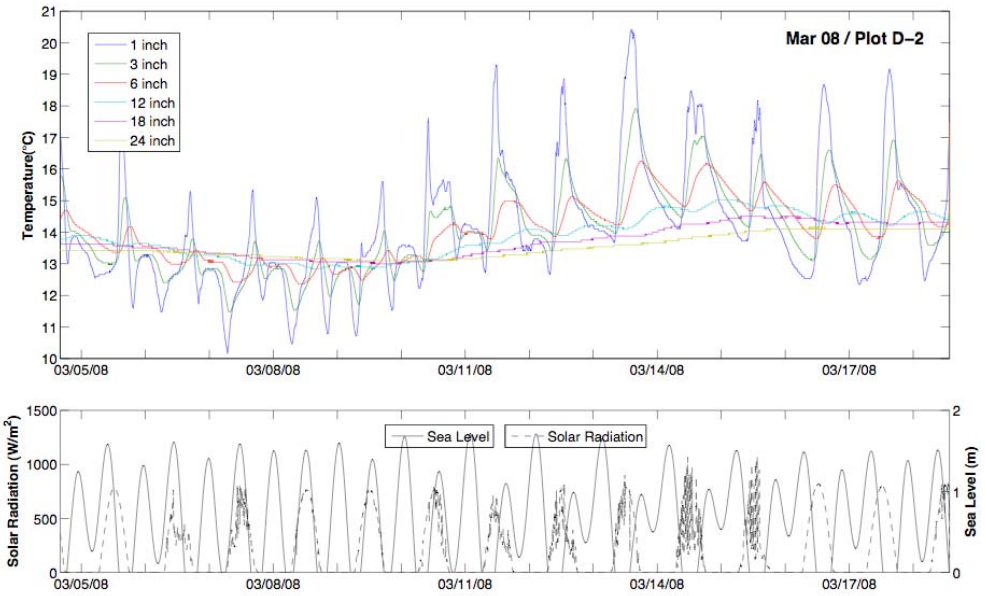

Figure 47. Temporal variations in the sediment temperatures (colored lines) and the sea level (black solid line) at Plot D-2, and the solar radiation (black dotted line) for 14 days in March 2008.

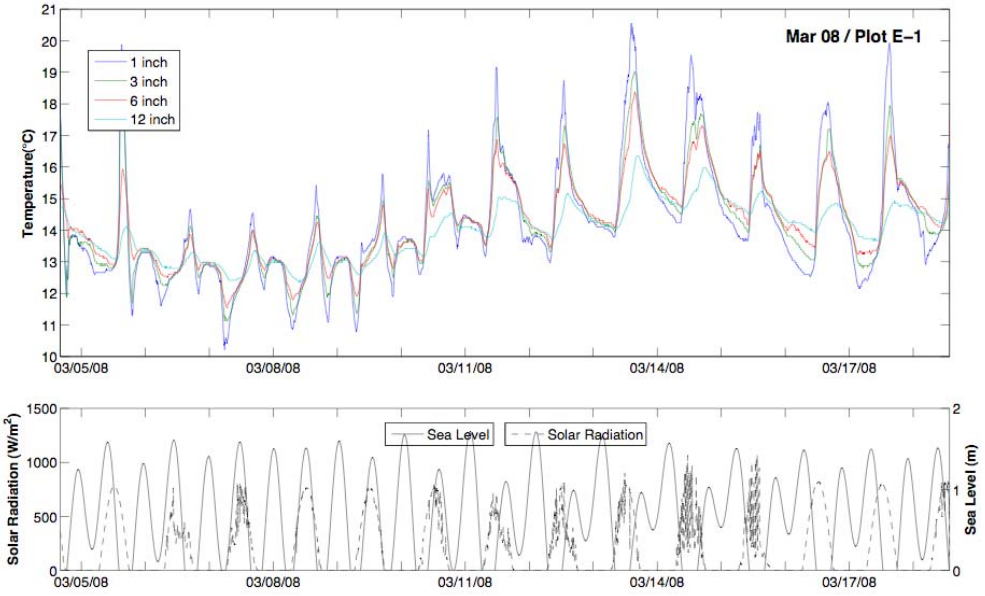

Figure 48. Temporal variations in the sediment temperatures (colored lines) and the sea level (black solid line) at Plot E-1, and the solar radiation (black dotted line) for 14 days in March 2008.

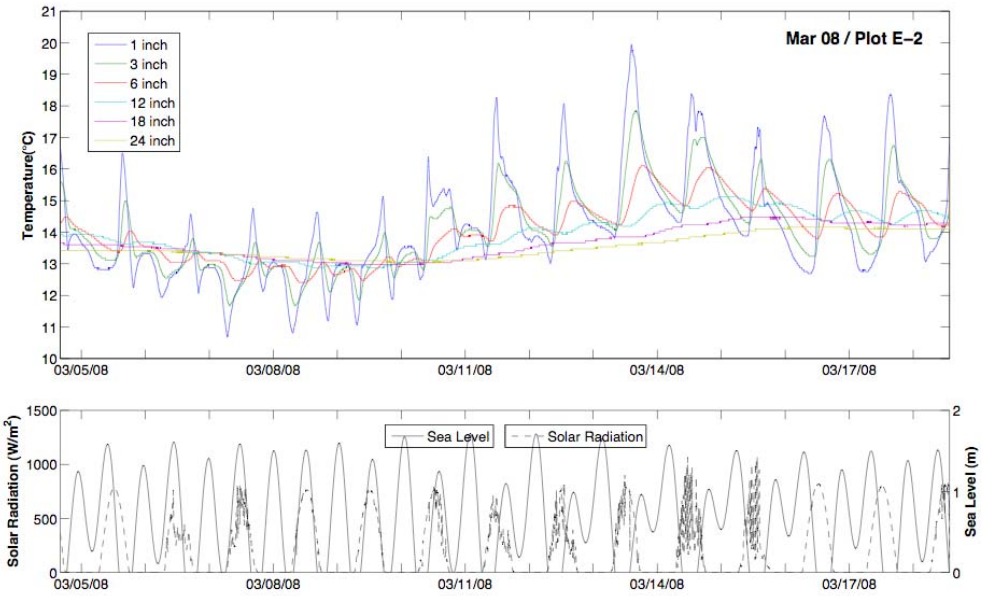

Figure 49. Temporal variations in the sediment temperatures (colored lines) and the sea level (black solid line) at Plot E-2, and the solar radiation (black dotted line) for 14 days in March 2008.

Anomalously large temperature variability was observed at depth (> 30 cm) in part of the control plot during both sampling periods (station E-2 August 2007 and station E-1 March 2008). Scatter plots and coefficients of determination confirmed the anomaly (

Table **8**, Table 9, and Figure 50). Temperature patterns from the two stations in the AC-treated plot D (D-1 and D-2) and one station in the non-mixed reference plot E (E-2) were similar, while temperature data for station E-1 showed weak correlation with the others. This difference between E-1 and the other sites was also observed at the other sampling depths, with the differences increasing with depth. Similarly, among the four August 2007 data sets, one logging station (E-2) from Plot E showed anomalous behavior compared to the other three temperature data sets (

Table **8).** The shape of these temperature profiles suggests a degree of thermal homogenization by macropore mixing around the loggers. It is possible that the seal between the instruments and surrounding sediments was compromised, permitting tidal flow and convective mixing within the profiles, but the confinement of this phenomenon to the undisturbed control plot suggests that the macropores may be of natural origin due to bioturbation or buried debris.

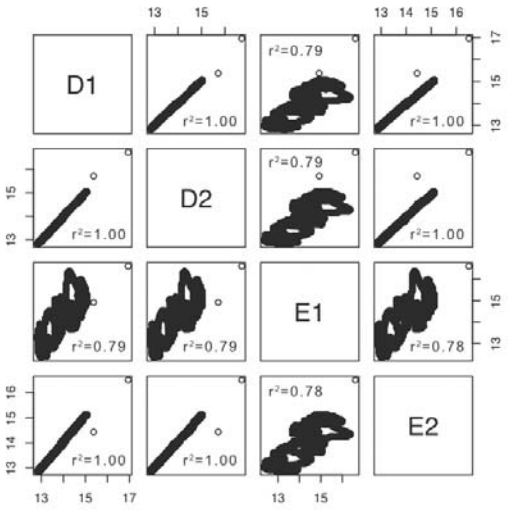

Figure 50. Scatter plots of temperature profiles of the four sampling locations at the depth of 12 inches (30 cm) from March 2008. The x and y axes represent sediment temperature (°C). Plot D is AC-amended; Plot E is undisturbed.

Table 8. Coefficients of determination among the four temperature profiles at various depths from the Aug 2007 sampling.

2.5 cm	D-1	D-2	E-1	E-2	5 cm	D-1	D-2	E-1	E-2
D-1	*	0.97	0.99	0.99	D-1	*	0.99	0.99	0.99
D-2	*	*	0.95	0.95	D-2	*	*	0.99	0.98
E-1	*	*	*	0.99	E-1	*	*	*	0.99
E-2	*	*	*	0.99	E-2	*	*	*	*
15 cm	D-1	D-2	E-1	E-2	30 cm	D-1	D-2	E-1	E-2
D-1	*	0.94	0.98	0.96	D-1	*	0.98	0.89	0.42
D-2	*	*	0.95	0.95	D-2	*	*	0.90	0.54
E-1	*	*	*	0.96	E-1	*	*	*	0.63
E-2	*	*	*	*	E-2	*	*	*	*

Table 9. Coefficients of determination among the four temperature profiles at various depths from the March 2008 sampling.

2.5 cm	D-1	D-2	E-1	E-2	5 cm	D-1	D-2	E-1	E-2
D-1	*	0.99	0.98	1.00	D-1	*	0.99	0.92	1.00
D-2	*	*	0.99	0.99	D-2	*	*	0.94	1.00
E-1	*	*	*	0.98	E-1	*	*	*	0.93
E-2	*	*	*	*	E-2	*	*	*	*
15 cm	D-1	D-2	E-1	E-2	30 cm	D-1	D-2	E-1	E-2
D-1	*	1.00	0.81	1.00	D-1	*	1.00	0.79	1.00
D-2	*	*	0.83	1.00	D-2	*	*	0.79	1.00
E-1	*	*	*	0.80	E-1	*	*	*	0.78
E-2	*	*	*	*	E-2	*	*	*	*

Heat transport model. **The sediment temperature profiles were simulated with either non-zero Darcy velocities (scenario one) or non-zero mechanical dispersion coefficients (scenario** two) during four days (day 2 - day 6) of the first week of sampling and four days (day 8 - day 12) of the second week of sampling in August 2007 and March 2008 (Table 10). Simulated temperature profiles with heat diffusion as the only transport process yielded RMSEs between the field data and the simulations much less than the precision of the loggers (0.5 °C) and close to the resolution (0.0625 °C) for most cases. For example, the simulation results for the first week of the March 2008 period at D-1 show that heat diffusion alone could explain the temperature fluctuation throughout the sediment profile with an RMSE value of 0.124 (Figure 51(A)).

Although heat transport by diffusion alone could largely account for the measured temperature fluctuations, still discrepancies remained. In some cases, the addition of an advection term enhanced the model fit. For the simulation results of week 1 of March 2008 at D-1, the least RMSE (0.049) was obtained by including a downward 10 cm/d Darcy velocity, less than one half of the RMSE without the advection term (Figure 51(B)). Depending on site and sampling time, the best fit Darcy velocity ranged from 1 cm/d to 10 cm/d (excluding E-1 in March 2008), and the reduction in RMSE compared to the diffusion-only simulations ranged from 0.3% to 60%.

It is also plausible that the saturated mudflat sediment may have no net fluid movement in the vertical direction. Instead, the sediments may experience oscillating flow due to tidal pumping, so a mechanical dispersion term was examined to assess whether the model fit could be enhanced by consideration of non-directional advective movement. As shown in

Table **10**, the use of mechanical dispersion in a limiting case with zero advection also enhanced the model fit compared to simulations with diffusion only, with a range of dispersion coefficients from 1×10^{-7} m^2/s to 1×10^{-6} m^2/s (excluding E-1 in March 2008). The enhancement in RMSEs compared to the diffusion-only cases ranged from 14% to 36%. An average dispersion coefficient for the mixed sediment plot D was 2.1×10^{-7} m^2/s.

As stated previously, some data from the unmixed Plot E (E-2 in August 2007 and E-1 in March 2008) significantly deviated from the others, showing penetration of large temperature fluctuations past 30 cm. The diffusion only RMSE values for these data ranged from 0.402 to 0.641, 2-7 times greater than for the other samples. The addition of mechanical dispersion yielded about 40% reduction in RMSE at E-1 in March 2008 (Figure 52), partially accounting for the possible macropore effects.

In summary, heat transport in the intertidal mudflat was explained mainly by heat diffusion. Advective pore water movement or mechanical dispersion helped explain the temperature profile in several cases, with Darcy velocities from 1 cm/d to 10 cm/d and mechanical dispersion coefficients of 1×10^{-6} m^2/s or less. The average best-fit Darcy velocity and mechanical dispersion coefficient for the AC mixed plot were 3.2 cm/d and 2.1×10^{-7} m^2/s, respectively.

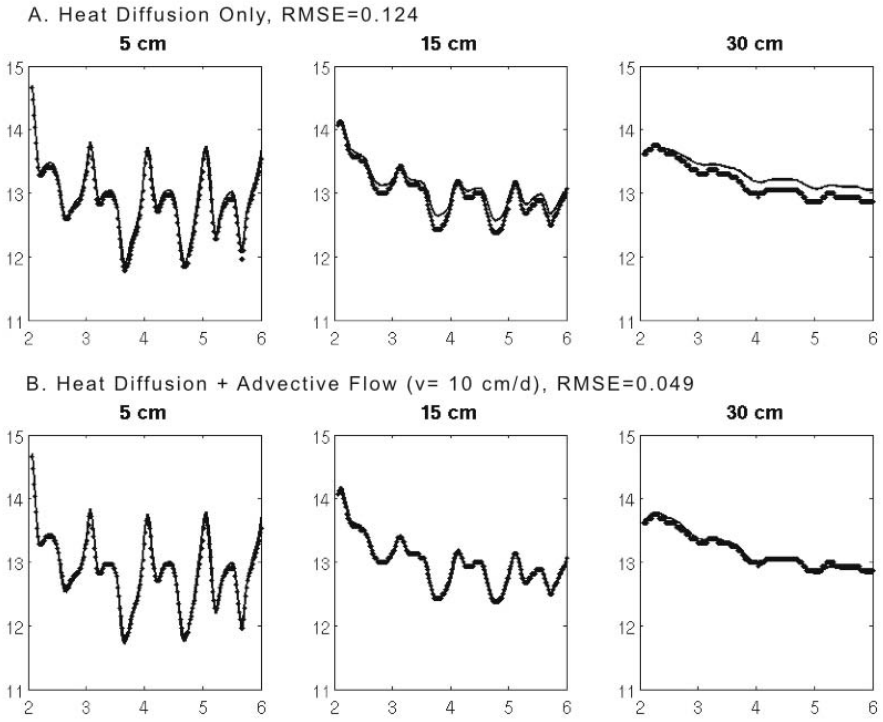

Figure 51. Simulation results for day 2 - day 6 (week 1) in March 2008 at Plot D-1. A) Heat diffusion only. B) Diffusion and advection. Measurements (dotted line) and simulation results (solid line). The x axes and y axes represent day and temperature (°C), respectively.

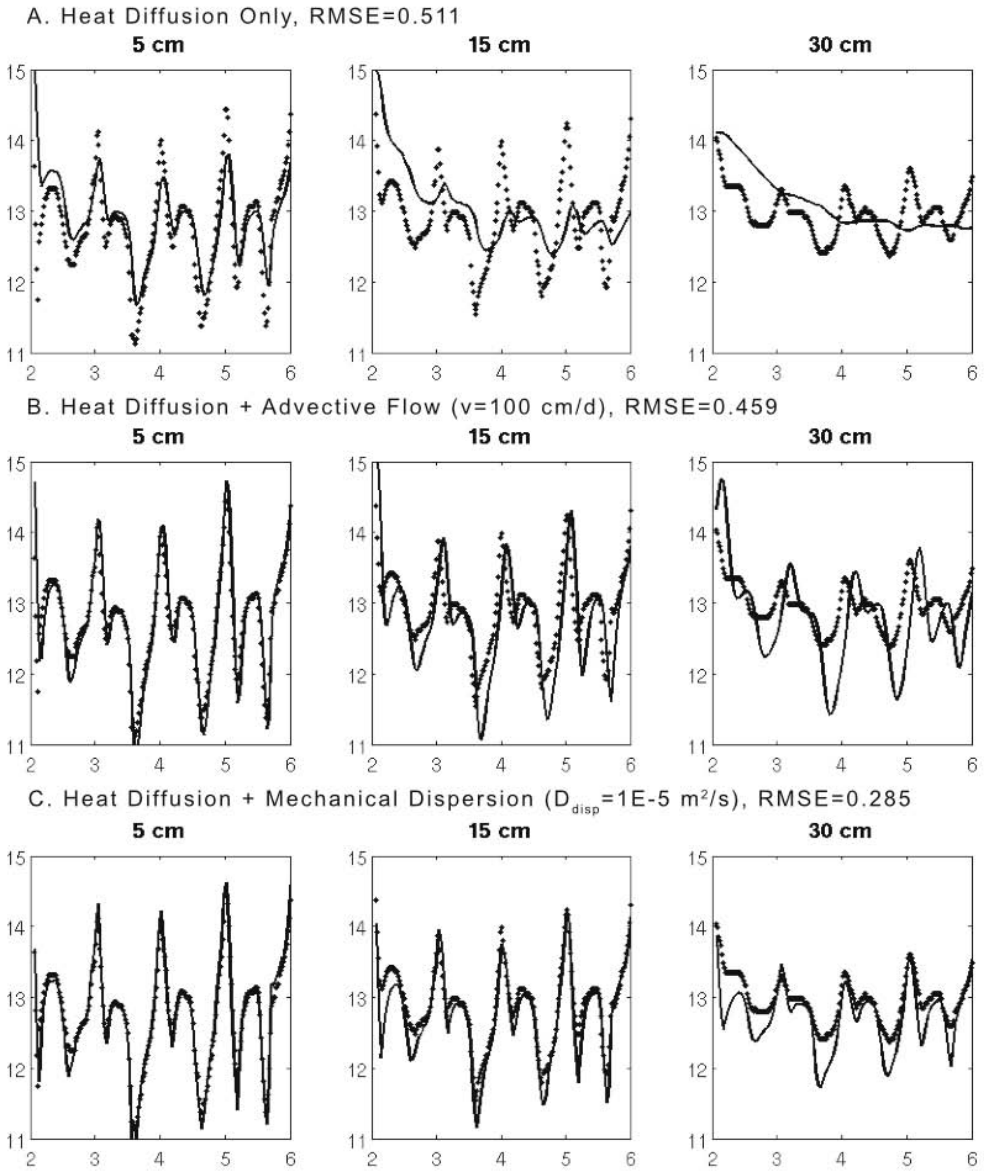

Figure 52. Simulation results for day 2 - day 6 (week 1) in March 2008 at Plot E-1. A) Heat diffusion only. B) Diffusion and advection. C) Diffusion and dispersion. Measurements (dotted line) and simulation results (solid line). The x axes and y axes represent day and temperature (°C), respectively.

Table 10. Heat transport model simulation summary considering diffusion only and addition of an advection or dispersion term. Best-fit models are presented for Darcy velocity (v), the dispersion coefficient (D_{disp}), and root mean squared error (RMSE).

Aug 2007	Plot	D-1	RMSE	D-2	RMSE	E-1	RMSE	E-2	RMSE
1st wk	heat diffusion only	-	0.093	-	0.166	-	0.203	-	0.402
	with advection v (cm/d)	10	0.073	1	0.163	10	0.143	10	0.346
	with dispersion D_{disp} (m^2/s)	1E-7	0.064	1E-7	0.137	1E-7	0.190	1E-7	0.385
2nd wk	heat diffusion only	-	0.093	-	0.258	-	0.123	-	0.641
	with advection v (cm/d)	1	0.083	1	0.257	1	0.115	10	0.547
	with dispersion D_{disp} (m^2/s)	1E-7	0.077	1E-6	0.190	1E-7	0.100	1E-6	0.564
Mar 2008	Plot	D-1	RMSE	D-2	RMSE	E-1	RMSE	E-2	RMSE
1st wk	heat diffusion only	-	0.124	-	0.100	-	0.511	-	0.108
	with advection v (cm/d)	10	0.049	1	0.088	100	0.459	10	0.060
	with dispersion D_{disp} (m^2/s)	1E-7	0.094	1E-7	0.074	1E-5	0.285	1E-7	0.071
2nd wk	heat diffusion only	-	0.119	-	0.114	-	0.588	-	0.172
	with advection v (cm/d)	1	0.094	1	0.088	1	0.585	1	0.139
	with dispersion D_{disp} (m^2/s)	1E-7	0.076	1E-7	0.097	1E-6	0.339	1E-7	0.124

Relative importance of mechanical dispersion for PCB mass transfer. To compare the relative contribution of advection and diffusion to PCB mass transfer, the dimensionless Peclet number (Pe = Lu/D) was evaluated, where L is the characteristic length (m), u is the interstitial velocity ([Darcy velocity]/[porosity], m/s), and D is a nominal diffusion coefficient for PCBs (5×10^{-10} m^2/s). Although PCB mass transfer would be retarded in the presence of AC by sorption, it would not affect the ratio because both transfer processes would be slowed equally.

An average Darcy velocity of v=3.2 cm/d and a porosity of 0.57 gave an average interstitial velocity of u=5.6 cm/d. The characteristic length was determined from the process of PCB stabilization by PCB mass transfer from sediment particles to AC particles. An average inter-particle distance between sediment particles and closest activated carbon particle was estimated assuming 3 wt% of homogeneous AC addition. For a simplified one-dimensional system the characteristic length estimate was L=200 μm. The resulting Peclet number is Pe=0.26, indicating that the PCB diffusion process contributes more than directional advective PCB transport to the sequestration of PCBs by AC, although interstitial flow should not be ignored.

Advective flow in a mudflat may vary temporally and spatially. From the heat transport simulations plausible interstitial flow rates up to 18 cm/d were obtained, implying that PCB transport by advective pore water flow could be more substantial at certain field conditions. The calculation of a Peclet number for PCBs or other pollutants is sensitive to the characteristic length estimate. The characteristic length in this case depends on the AC distribution, AC dose, and other non AC-related factors. For example, if the AC distribution is not homogeneous, then the characteristic length for mass transfer would be large. If the AC dose is increased, the distance between AC and sediment particles will decrease. Further study is needed to examine these factors in consideration of field data.

Moreover, PCB mass transfer by mechanical dispersion should be acknowledged, which is driven by advective pore water movement. From the limiting scenario of zero

net directional advective movement but non-zero mechanical dispersion, the contribution of mechanical dispersion was evaluated by comparison to the molecular diffusion. The average dispersion:diffusion coefficient ratio was on the order of 400:1. This suggests that mechanical dispersion may accelerate PCB mass transfer compared to molecular diffusion alone. As discussed for the advection scenario, this mechanical dispersion effect would also vary depending on field conditions, and the dispersion/diffusion ratio might range from several hundreds to thousands.

The work on mass transfer modeling has been accepted with revisions to Environmental Science and Technology by Cho et al. in 2010 [(Cho, Moffett et al.), Appendix A].

VI. Conclusion and Implications for Future Research

Establishing correlations between physiochemical measurement tools (e.g., PEDs and POM samplers) and bioavailability in the field will allow members of the scientific community, sediment managers and DoD users to confidently use these rapid, inexpensive tools as an additional line of evidence to assess trends in contaminant availability affecting the ecological recovery of a contaminated sediment site after treatment and during a monitored recovery. A conceptual framework is being developed to predict ecosystem recovery, given some knowledge of chemical properties of the sediment, basic information about biodynamics for the contaminants of interest, and benthic community data of reference sites in the recruitment pool. The conceptual framework integrates biodynamic modeling to assess the required clean-up levels that enable recovery. The framework helps to estimate the required reduction in bioavailability (assimilation efficiency) that is necessary to reduce bioaccumulation to levels observed at references sites (ambient pollution levels). First, the site-specific correlation between bioavailability and contaminant availability to pore water has to be established with dual deployments of bioassays and passive samplers. Then, sediment managers and DoD users can use passive samplers as a surrogate for contaminant availability, which then allows for greater spatial and temporal data for large-scale risk assessment for the entire site. With the knowledge of correlations to bioavailability and information concerning the benthic organism recruitment pool the extent of ecosystem recovery that may follow remediation can be estimated. The research has been successfully completed in regards to the project milestones as summarized below.

1. Demonstrate the use of rapid assessment tools in the field and correlate with conventional methods

1.1 Characterize (in the laboratory) relevant parameters for the use of polyethylene sampling devices

Sediment pore water concentrations of polychlorinated biphenyls (PCBs) in a contaminated mudflat in San Francisco Bay, CA, were determined by field-deployed polyethylene devices (PEDs). Sequential sampling of PEDs deployed in the field sediment showed large differences in uptake rates and time to equilibrium compared to PEDs mixed with field-collected sediment in the laboratory. A modeling approach that involves the use of impregnated performance reference compounds (PRCs) has ben demonstrated and interpretation of the data either by PCB molar volume adjustment or environmental adjustment factors to measure pore water concentrations of 118 PCB congeners. Both adjustment methods predicted comparable sampling rates, and PCB pore water concentrations estimated by use of the molar volume adjustment method were similar to values analytically measured in pore waters from the laboratory and field. The utility of PEDs for sampling pore water in the field was evaluated at a tidal mudflat amended with AC to sequester PCBs. Pore water concentrations decreased up to 60% within 18 months after activated carbon amendment, as compared to a mechanical-mixed control plot. Results of this study illustrate that PEDs provide an inexpensive, in-situ method to measure total PCB contamination in sediment pore water.

The milestones for PED passive samplers to measure pore water concentration in sediment have been satisfied. Further details can be found in the publication of the PED work in *Environmental Science and Engineering* under Tomaszewski and Luthy (2008) (Appendix A).

1.2 Apply PEDs and PCB immunoassay in the field at treated and untreated sites

An analytical protocol for the immunoassay test on Hunters Point sediment samples was established. The PCB concentrations from sediment samples measured by the immunoassay tests were reported as Aroclor 1254 and the result was compared with the result obtained from GC-based analysis. Reasonable correlation was found. The immunoassay captures the effects of PCB sequestration after carbon amendment because PCBs sorbed on activated carbon become less available for the soft methanol extraction. When the immunoassay data were compared with pore water concentrations measured by PEDs, a noticeable correlation with pore water concentrations could only be observed for sediment samples from the top 6 inches where PEDs where deployed. The immunoassay method is semi-quantitative and, at best, useful for assessing range values of total PCBs in sediment.

1.3 Quickly and effectively measure pore water PCB concentration in the field

The effectiveness of activated carbon amendment to sequester PCBs in sediment was measured by deploying PEDs in four treatment plots. Pore water concentrations decreased up to 60% within 18 months after activated carbon amendment, as compared to a mechanical-mixed control plot.

Initial results from the in-situ measurement of vertical profiles with passive samplers in sediment are presented. The significance of these results will be further explored with regard to desorption kinetics of the PRCs and the uptake kinetics of the PCBs in order to calculate pore water concentrations. Furthermore, the usefulness of passive samplers to measure in-situ pore water concentrations as a rapid and predictive measurement of contaminant availability in sediments will be evaluated, which is greatly needed for risk assessment and remedial treatment evaluation in the field. This work will continue in the future during Phase II with column studies and PEDs to validate a mass transfer model for PCBs and AC under minimally mixed conditions. A manuscript about the work of measuring vertical pore water with the title “Measurement of in-situ PCB pore water concentration profiles in AC-amended sediments using passive samplers” is currently in preparation (Oen, Janssen et al.).

Further details can be found in the publication of the in-situ PED work in *Environmental Science and Technology* under Tomaszewski and Luthy (2008) (Appendix A).

1.4 Demonstrate that PCB pore water concentrations are indicators of mass transfer to activated carbon particles

Measurements of reduced pore water concentration in combination with reduced bioavailability proved to be good indicators for assessing the extent of mass transfer of PCBs to AC particles. Laboratory studies under well-mixed conditions are preferred to assess in comparatively short time the potential benefits of carbon dose and particle size on treatment effectiveness. In-situ pore water measurements are important to assess the

extent of sequestration for a particular field deployment method (mixing regime etc.). The conducted bioassays were relevant to reflect the residual bioavailability of PCBs in the presence of AC particles, which varies among species depending on their feeding strategy. That is, depending on whether the primary route of uptake for an organism is from the aqueous phase, or by particle ingestion and what types of particles are ingested.

2. Conduct regional surveys to determine the benthic species recruitment pool for Hunters Point

2.1 Define recruitment pool of benthic species, along with model and data needs

An analysis based on functional traits of feeding, reproduction, and position in the sediment shows that Hunters Point is depauperate in deposit feeders, subsurface carnivores, egg laying species, and species with no protective barrier. These species – which consume the sediment or prey on species that consume the sediment, lay their eggs on the sediment, and freely burrow through the sediment – are expected to become members of the benthic community at Hunters Point when the ecosystem recovers. This method of examining the benthic communities also identified the species which are not good functional fits for a contaminated environment and this approach appears to be a meaningful direction for future research.

A technical manuscript is in preparation by Janet Thompson et al. titled "Using functional ecology to examine pollution effects on benthic communities: examples with a long term study and a spatially intensive study" (Thompson, Luoma et al.).

3. Biodynamic modeling to predict PCB uptake by benthic organism

3.1 Conduct laboratory experiments to define physiological coefficients for biodynamic model

This work demonstrates that activated carbon amendment reduces the bioavailability of PCBs to benthic invertebrates (i.e., the marine clam *Macoma balthica*, the freshwater clam *Corbicula fluminea,* the marine polychaete *Neanthes arenaceodentata*, and the freshwater oligochaete *Lumbriculus variegates*) under laboratory conditions by 85 to 95%. These studies support the potential for activated carbon amendment as a sediment remediation strategy in both freshwater and marine systems. The biodynamic model presented herein offers promise as a predictive tool for engineers and practitioners. The model was successfully parameterized and tested for these four organisms to predict PCB tissue concentrations and to quantify uptake over different exposure routes of sediment ingestions and water. As long as species-specific input parameters can be obtained or measured, sediment-specific parameters may be estimated and adjusted to describe potential bioavailability at a prospective treatment site. Measures of reductions in aqueous equilibrium PCB concentrations can be used to estimate the proportional mass transfer of PCBs from native sediment particles to the added carbon. Use of sediment-ingesting benthic organisms in model calculations will provide the best indicator of local bioavailability. Further details of the work on

bioaccumulation and modeling can be found in *Environmental Science and Technology* under McLeod et al. (2008) and Janssen et al. (2010) (Appendix A).

The results from Phase I of this SERDP project (ER-1552) and occasional concerns about possible adverse biological effects of carbon material suggest further investigations. In Phase II of this project, exposure experiments will be designed to resolve whether adverse affects to benthic invertebrates result from sorbent addition to the sediment considering ingestible and non-ingestible particle sizes for the amendment.

3.2 Demonstrate applicability of biodynamic model in the field

AC-amendment significantly reduced bioaccumulation under field and laboratory conditions. The study revealed three factors that influence in-situ bioassays: 1. Deposit-feeders exhibit less PCB uptake from untreated sediment when feeding is reduced, which is probably related to higher food quality in the field versus the laboratory; 2. AC-amendment significantly reduces bioavailability under laboratory and field conditions; and 3. Sediment deposition within test cages in the field partially masks the remedial benefit of underlying AC-amended sediment. Ex-situ and in-situ experiments inevitably show some differences that are associated with measurement methods and effects of the environment. Parallel ex-situ and in-situ bioassays and passive sampler measurements can tease apart these field influences and the biodynamic model helps confirm and quantify influences on PCB exposure. Additional analysis of food availability and in-situ ingestion rates should be considered in future studies.

A technical manuscript titled "Assessment of field-related influences on polychlorinated biphenyl exposures and sorbent amendment using polychaete bioassays and passive sampler measurements" has been submitted to *Environmental Toxicology and Chemistry* in May 2010 (Janssen, Oen et al. 2010).

4. Development and application of a predictive general ecosystem recovery model

4.1 Predict ecological recovery and compare to field studies

Sediment chemistry analysis shows that PCBs are the major risk drivers at Hunters Point (1570 ppb) and that the reference sites contain very low levels of PCB contamination (9 ppb). Different feeding traits present a direct pathway of exposure, which can be mechanistically linked to PCB bioaccumulation by biodynamic modeling. The model shows that the deposit feeder *Neanthes arenaceodentata* accumulates about 20-times more PCBs in its lipids than the facultative deposit feeder *Macoma balthica* and up to 130-times more than the filter feeder *Mytilus edulis*. The comparison of different exposure scenarios suggests that PCB tissue concentrations at Hunters Point are two orders of magnitude higher than at the reference sites. With application at full-scale, an in-situ sorbent amendment with activated carbon may reduce PCB bioaccumulation at Hunters Point by up to 85 to 90% under favorable field and treatment conditions. The modeling framework further demonstrates that such expected remedial success corresponds to exposure conditions suggested as the cleanup goal for Hunters Point. However, concentrations remain slightly higher than at the reference sites. The present study demonstrates how the remedial success of a sorbent amendment, which lowers the

PCB availability, can be compared to reference conditions and traditional cleanup goals, which are commonly based on bulk sediment concentrations.

A technical manuscript is in preparation titled "PCB-induced changes of the benthic community composition and predictions of ecosystem recovery following in-situ sorbent amendment for Hunters Point, California" (Janssen, Thompson et al.).

4.2 Correlate conventional alternative assessment methods and model ecosystem recovery in the field

On average, the number of species and the range in abundance of species at the Hunters Point and at the reference sites did not differ greatly. Given the similar diversity index between the reference sites and Hunters Point, it may be concluded that the benthic communities did not differ greatly. Non-metric multi-dimensional scaling (MDS) plots show that the benthic communities of the intertidal reference sites are more similar to the other reference sites than to Hunters Point. In examining the functional groups of these sites, significant differences between the reference sites and the Hunters Point site were found.

Diversity indices and the abundance of indicator species were not sensitive enough to recognize the changed community composition at Hunters Point. The analysis on the basis of functional ecology identified the lack of deposit feeders, the absence of egg laying species and lack of species with no barrier or a weak barrier at Hunters Point relative to the 30 reference sites. The comparison of functional ecology resulted in the most meaningful analysis approach to identify changes in the community composition for this study.

5. Monitor and assess carbon amendment in Grasse River

5.1 Bioaccumulation studies with Lumbriculus variegates

A marked reduction in PCB uptake by the aquatic oligochaete *Lumbriculus variegatus* from Grasse River sediment amended with 2.5 % (dry wt) activated carbon was documented. Further, a biodynamic model successfully predicted total PCB uptake over a 28-day time series for worms exposed to un-amended and activated carbon amended sediment. The biodynamic model was also validated for a freshwater clam with sediment from the Grasse River and a marine clam with sediment from Hunters Point. This was published as a paper in *Environmental Science and Engineering* [Appendix A, (McLeod, Luoma et al. 2008)].

5.2 PCB mass transfer modeling with heterogeneous mixing

In this study, the relative significance of molecular diffusion, directional advective flow, and mechanical dispersion for mass transfer of a PCB, a typical example of hydrophobic organic compounds (HOCs), was investigated in sediment. While the information is useful to assess general aspects of HOC transport and fate, it is especially useful to predict the long-term benefit of *in-situ* sequestration by activated carbon (AC)

sorbent. The existence of advective flow and mechanical dispersion would accelerate the remedial action by AC and shorten the monitoring period, and thus likely further increase public acceptance of the remediation option. Heat transport modeling is a convenient and applicable indirect method to examine such flow and dispersion for shallow mudflat or marsh-like sediments. It is noted that this method has certain lower-bound quantification limits for both Darcy velocities and dispersion coefficients due to the faster process of heat diffusion than molecular diffusion for HOCs in many cases. So, the model may be unable to extract values from a system with quite small, but still-significant, advective flow or mechanical dispersion less than the detectable limits. However, these caveats did not affect the utility of the method in this study because the simulation results showed significantly higher values of Darcy velocity (3.2 cm/d) and mechanical dispersion (2.1×10^{-7} m^2/s) than the estimated detection limits (0.8 cm/d and 5.6×10^{-9} m^2/s). This information and modeling is expected to enable more reliable assessments of the fate and movement of HOCs in sediments and the benefit of AC amendment in the aquatic environment in future research.

This study provides one of the very few data sets and modeling of pore water movement in the upper, biologically active layer for shallow, intertidal mudflat sediments. The work on mass transfer modeling has been accepted with revisions to Environmental Science and Technology by Cho et al. in 2010 [(Cho, Moffett et al.), Appendix A].

Our recent ESTCP field project and the Phase I of the SERDP project (ER-1552) show the need for predictive models to assess the long-term performance of AC amendment under quiescent field conditions and slow mass transfer compared to well-mixed conditions in the laboratory. Previous fieldwork demonstrated that sediments in a contaminated tidal mudflat can be amended with AC using commercial equipment and thereby reduce exposures to pore water and benthic organisms. It was shown that AC in the field retained its capacity to continually sorbed PCBs months after deployment. However, less overall reductions in the field versus the laboratory calls for predictive models to assess long-term trends in PCB-pore water concentrations and availability under field conditions with slow mass transfer and heterogeneous AC distribution, as may result from brief mixing events in the field.

Among novel sorbents considered for *in situ* treatment of sediments, AC is the most promising to gain regulatory approval for actual field deployment. AC field deployment tests were completed at Hunters Point in San Francisco Bay and the Grasse River in upstate New York. Additional laboratory and field studies are underway at other sites. The AC itself is relatively benign and mimics naturally-occurring black carbon material like char or charcoal.

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VIII. Appendix

(A) List of Technical Publications

1. McLeod, P. B., S. N. Luoma, et al. (2008). "Biodynamic modeling of PCB uptake by Macoma balthica and Corbicula fluminea from sediment amended with activated carbon." *Environmental Science & Technology* 42(2): 484-490.

2. Tomaszewski, J. E. and R. G. Luthy (2008). Field deployment of polyethylene devices to measure PCB concentrations in pore water of contaminated sediment. *Environmental Science & Technology* 42(16): 6086-6091.

3. Janssen, E. M.-L.; Croteau, M.-N.; Luoma, S. N.; Luthy, R. G. (2010). Measurement and modeling of polychlorinated biphenyl bioaccumulation from sediment for the marine polychaete *Neanthes arenaceodentata* and response to sorbent amendment. *Environmental Science and Technology*, 44, 2857-2863.Cho, Y.-M.; Werner, D.; Moffett, K. B.; Luthy, R. G. (2010). "Assessment of advective pore water movement affecting mass transfer of hydrophobic organic contaminants in marine in*tertidal sediment"*. *Environmental Science & Technology* 14 (*15), 5842-8.*

4. Janssen, E. M.-L.; Oen, A. M. P.; Luoma, S. N.; Luthy, R. G.(2011). "Assessment of field-related influences on polychlorinated biphenyl exposures and sorbent amendment using polychaete bioassays and passive sampler measurement". Submitted to *Environmental Chemistry and Toxicology*, 30 (1). 173-180.

5. Hale S. E., Tomaszewski J. E., Werner D., Luthy R. G. (2009). Sorption of dichloro-diphenyl-trichloroethane (DDT) and its metabolites by activated carbon in clean water and sediment slurries. *Water Research*, 43, 4336-4346.

6. Werner D., Hale S. E., Ghosh U., Luthy R.G. (2010). Polychlorinated Biphenyl Sorption and Availability in Field-contaminated Sediments. *Environ. Sci. Technol*, 44, 2809-2815.

Submitted manuscripts

Janssen, E. M.-L.; Thompson, J.K.; Luoma, S. N.; Luthy, R. G. "PCB-induced changes of a benthic community and expected ecosystem recovery following in-situ sorbent amendment", submitted to *Environmental Chemistry and Toxicology*, December 2010.

Manuscripts in preparation

1. Oen, A. M.P.; Janssen, E. M.-L.; Erk, E.; Breedveld, G.D.; Cornelissen, G.; Luthy, R.G. "Measurement of in-situ PCB pore water concentration profiles in AC-amended sediments using passive samplers".

2. Thompson, J.K.; Luoma, S.N.; Shouse, M.K.; Parchaso, F. "Using functional ecology to examine pollution effects on benthic communities: examples with a long term study and a spatially intensive study".

Conference or symposium proceedings

1. **Northern California Society of Environmental Toxicology and Chemistry Annual Meeting**, Berkeley, CA, USA, May 2007. Tomaszewski, J. E.; Luthy, R.G. "Combining Passive Samplers and Bioassays to Assess the Effectiveness of Activated Carbon Amendment of Contaminated Sediment".

2. **SETAC Conference, North America**, Tampa, FL, 16/11-20/11 2008. Janssen, E. M.-L.; Croteau, M.-N. Luoma, S. N.; Luthy, R.G. "Bioaccumulation of PCBs in sediment by the polychaete Neanthes arenaceodentata and effects of activated carbon amendment".

3. **SERDP: Partners in Environmental Technology Technical Symposium & Workshop,** Washington, D.C., 12/2-12/5 2008. Luthy, R.G. "Application and Assessment of Amendments at Sediment Sites".

4. **Workshop on New Advances in Sediment Remediation** at the University of Maryland Baltimore County (UMBC), 14/11/2008. Janssen, E. M.-L.; Croteau, M.-N. Luoma, S. N.; Luthy, R.G. "Bioaccumulation of PCBs in water by the polychaete N. arenaceodentata".

5. **International Network for Sediment Research (INSR)**, at University of Newcastle, UK, Workshop 29-30/05 2009. Janssen, E. M.-L.; Croteau, M.-N. Luoma, S. N.; Luthy, R.G. "Measurement and Modeling of PCB bioaccumulation in marine polychaete and effects of sorbent amendment".

6. **SERDP: Partners in Environmental Technology Technical Symposium & Workshop,** Washington, D.C., 12/3-12/6 2009; Thompson, J.K.; Luoma S.N., Luthy R.G. "Assessing ecological recovery by functional ecology".

7. **SETAC Conference, Europe**, Seville, Spain, 24/05-27/05 2010. Janssen, E. M.-L.; Oen, A. M.P.; Thompson, J.K.; Luoma, S. N.; Luthy, R.G. **"**In-situ measurement and modeling of PCB bioaccumulation and ecosystem recovery".

8. **American Chemical Society 237th Annual Meeting,** Division of Environmental Chemistry, Symposium in Honor of James O. Leckie, Salt Lake City, UT, March 22-26, 2009. Richard G. Luthy, Yeo-Myoung Cho, Upal Ghosh, Alan J. Kennedy and Todd S. Bridges. "Field Application of Activated Carbon Amendment for In-situ Stabilization of PCBs in Sediment".

9. **Invited keynote presentation, 5th International SedNet Conference**, Urban Sediment Management and Port Redevelopment & Sediment in River Basin Management Plans, Oslo, Norway, May 27-29, 2007**. Luthy, R.G.** "In-place treatment of persistent organic contaminants in sediments through addition of activated carbon".

Further oral presentations

1. **Natural History Museum,** London, UK, 07/12/2009. Janssen, E. M.-L.; Oen, A. M.P.; Thompson, J.K.; Luoma, S. N.; Luthy, R.G. "Measurement and modeling of bioaccumulation and ecosystem recovery after in-situ sediment treatment".

2. **University Tübingen,** Applied Geoscience, Germany, 27/11/2009. Janssen, E. M.-L.; Oen, A. M.P.; Thompson, J.K.; Luoma, S. N.; Luthy, R.G. "Measurement and modeling of bioaccumulation and ecosystem recovery after in-situ sediment treatment".

3. **University of Newcastle Upon Tyne,** Civil Engineering and Geosciences, Newcastle Upon Tyne, UK, 20/11/2009. Janssen, E. M.-L.; Oen, A. M.P.; Thompson, J.K.; Luoma, S. N.; Luthy, R.G. "Measurement and modeling of bioaccumulation and ecosystem recovery after in-situ sediment treatment".

4. **Swiss Federal Institute of Aquatic Science and Technology [Eawag] & ETH, Zurich, and Ecole Polytechnique Federale de Lausanne, Switzerland,** September 2010, R. G. Luthy, **"**Laboratory and Field Evaluation of Amendment for In-situ Stabilization of Organic Pollutants in Sediments."

5. **Tsuan Hua Feng Distinguished Lecture in Environmental Engineering**, University of Massachusetts, September 27, 2007. Luthy, R. G. "Cleaning up sediments without throwing the mud".

6. **General Electric Global Research**, invited presentation, Environmental Technology Laboratory, Niskayuna, NY, November 19, 2007.Luthy, R.G. "Cleaning up sediments without throwing the mud".

7. **Inaugural CH2M-Hill Distinguished Lecture**, Department of Civil and Environmental Engineering, Virginia Tech, Blacksburg, VA, April 7, 2008. Luthy, R.G. "Cleaning up sediments without throwing the mud."

8. **Sixth International Conference on Remediation of Chlorinated and Recalcitrant Compounds,** Monterey, CA, May 19-22, 2008, invited presentation, Luthy, R.G.; Y-M Cho; A.J. Kennedy; T.S. Bridges; U. Ghosh. "Field Testing of Activated Carbon Mixing and In Situ Stabilization of PCBs in Sediment".

Conference poster presentations

Poster title: "Measurement and Modeling of Ecosystem Risk and Recovery for In Situ Treatment of Contaminated Sediment". Janssen, E. M.-L.; Cho, Y.-M.; Tomaszewski, J. E.; Oen, A. M.P.; Thompson, J. K.; Luoma, S. N.; Luthy, R. G. presented at:

1. International Network for Sediment Research (INSR), at University of Newcastle, UK, Workshop 05-06/07 2010
2. Oceans Colloquium, 4/23/ 2010, Hopkins Marine Station, Pacific Grove, CA
3. SERDP: Partners in Environmental Technology Technical Symposium & Workshop, Washington, D.C., 12/3-12/6 2009.
4. Sediment network (SedNet) conference, Hamburg, Germany, 10/7-10/9 2009.
5. International Network for Sediment Research (INSR), at University of Newcastle, UK, Workshop 29-30/05 2009
6. SERDP: Partners in Environmental Technology Technical Symposium & Workshop, Washington, D.C., 12/2-12/5 2008
7. SERDP and ESTCP Expert Panel Workshop on Research and Development Needs for Understanding and Assessing the Bioavailability of Contaminants in Soils and Sediments, Annapolis, MD, 8/20/ 2008
8. Gordon Research Conference, Holderness School in Holderness NH United States, 06/22- 06/27 2008
9. SERDP: Partners in Environmental Technology Technical Symposium & Workshop, Washington, D.C., 12/4-12/6 2007

(B) Supporting data

1. Sample locations for benthic surveys
2. Species and functional groups
3. Information about the chemical data stations
   - Sample identification, location, and depth
   - Physical properties and TOC content
   - Sediment chemistry for the selected stations in the Central Bay available from the SFEI data base
   - Additional stations for TOC and total PCB concentrations in sediment provided by the SFEI but not available from the online data base
4. Total abundance data
5. Total and normalized abundance plots
6. Data for Figure 31
7. Data for Figure 34
8. Date for Figure 35
9. Data for Figure 28

1. Sample locations for benthic surveys

Biological Information									
study	side code	latitude	longitude	depth [m]		study	side code	latitude	longitude	depth [m]
#1 (Candlestick)						#3 (Emeryville)				
USGS	BS 33	37.7044	-122.3852	0.9		USGS	BS 17	37.8451	-122.3523	3.0
USGS	BS 34	37.7045	-122.3796	1.2		USGS	BS 18	37.8494	-122.3398	2.4
USGS	BS 35	37.7045	-122.3743	1.5		USGS	BS 19	37.8540	-122.3271	2.1
USGS	BS 36	37.7045	-122.3694	1.5		USGS	BS 20	37.8574	-122.3178	1.5
USGS	BS 37	37.6953	-122.3846	1.2		#4 (Berkeley / Richmond)				
USGS	BS 38	37.6953	-122.3794	1.2		USGS	BS 5	37.8844	-122.3765	1.8
USGS	BS 39	37.6954	-122.3742	1.2		USGS	BS 6	37.8883	-122.3634	1.5
USGS	BS 40	37.6953	-122.3698	1.5		USGS	BS 7	37.8923	-122.3518	1.5
USGS	BS 41	37.6862	-122.3840	0.9		USGS	BS 8	37.8956	-122.3383	1.5
USGS	BS 42	37.6861	-122.3792	0.9		USGS	BS 9	37.8759	-122.3707	1.8
USGS	BS 43	37.6860	-122.3744	1.2		USGS	BS 10	37.8803	-122.3587	1.8
USGS	BS 44	37.6860	-122.3699	1.2		USGS	BS 11	37.8855	-122.3461	1.8
USGS	BS 45	37.6794	-122.3797	0.9		USGS	BS 12	37.8895	-122.3333	1.8
USGS	BS 46	37.6794	-122.3746	1.2		USGS	BS 13	37.8686	-122.3659	2.4
USGS	BS 47	37.6793	-122.3698	1.5		USGS	BS 14	37.8727	-122.3542	1.8
USGS	BS 48	37.7046	-122.3918	0.0		USGS	BS 15	37.8781	-122.3418	1.8
USGS	BS 49	37.6952	-122.3908	0.0		USGS	BS 16	37.8828	-122.3293	1.5
USGS	BS 50	37.6863	-122.3888	0.0		#6 (Tiburon)				
USGS	BS 51	37.6793	-122.3873	0.0		USGS	BS 1	37.9082	-122.4701	1.5
#2 (Alameda)						USGS	BS 2	37.9009	-122.4615	2.4
USGS	BS 24	37.7654	-122.3030	2.4		USGS	BS 3	37.8975	-122.4567	3.0
USGS	BS 25	37.7635	-122.2962	1.8		USGS	BS 4	37.8975	-122.4567	3.0
USGS	BS 26	37.7615	-122.2896	2.1			Hunters Point			
USGS	BS 27	37.7588	-122.2837	1.5		USGS	SB-16	37.7228	-122.3768	0.0
USGS	BS 28	37.7562	-122.2794	1.2						
USGS	BS 29	37.7532	-122.2750	0.9						
USGS	BS 30	37.7602	-122.2991	2.4						
USGS	BS 31	37.7581	-122.2928	2.4						
USGS	BS 32	37.7556	-122.2869	1.5						
USGS	BS 52	37.7619	-122.2732	0.0						
USGS	BS 53	37.7596	-122.2680	0.0						
	intertidal									
USGS	United States Geological Survey benthic surveys 2007									

2. Species and functional groups

TAXON	Feeding Group	Reproductive Group	Barrier Group
Cnidaria			
Hydrozoa	Filter Feeder	Broadcast spawn or fission with pelagic larvae	
Anthozoa	Filter Feeder	Broadcast spawn or fission with pelagic larvae	
Actiniaria	Filter Feeder	Broadcast spawn or fission with pelagic larvae	No barrier
Burrowing anenome (#1)	Filter Feeder	Broadcast spawn or fission with pelagic larvae	Tubed with tissue
Attached (#2)	Filter Feeder	Broadcast spawn or fission with pelagic larvae	No barrier
Edwardsiidae (?Scolanthus spp)	Filter Feeder	Broadcast spawn or fission with pelagic larvae	Tubed with tissue
Diadumene spp.	Filter Feeder	Broadcast spawn or fission with pelagic larvae	No barrier
Unid. Actiniaria (Diadumene?)	Filter Feeder	Broadcast spawn or fission with pelagic larvae	No barrier
Alcyonacea - Pennatulidae	Filter Feeder	Broadcast spawn or fission with pelagic larvae	No barrier
Stylatula spp.	Filter Feeder	Broadcast spawn or fission with pelagic larvae	No barrier
Turbellaria	Surface Carnivore		No barrier
Nemertea	Surface Carnivore	Viviparous/oviparous with surface juvenile	No barrier
Tubulanus spp.	Surface Carnivore	Broadcast spawn with pelagic larvae	No barrier
Unidentified Nemertea	Surface Carnivore	Viviparous/oviparous with surface juvenile	No barrier
Nematoda	Parasite/Deposit or Carnivore	Viviparous/oviparous with surface juvenile	No barrier
Phoronida	Filter Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Phoronis spp.	Filter Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Annelida			
Oligochaeta	Subsurface Deposit Feeder	Viviparous/oviparous with surface juvenile	
Naididae	Subsurface Deposit Feeder	Egg layer to the surface and asexual	No barrier
Tubificidae	Subsurface Deposit Feeder	Viviparous/oviparous with surface juvenile	No barrier
Tubificoides wasselli	Subsurface Deposit Feeder	Viviparous/oviparous with surface juvenile	No barrier
Polychaeta			
Tenonia priops	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Harmothoe imbricata	Surface Deposit Feeder	Egg layers with pelagic larvae	No barrier
Malmgreniella macginintiei	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Malmgreniella spp. (= Harmothoe spp.)	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Pholoe spp.	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Exogone lourei	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	No barrier
Exogone spp.	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	No barrier
Sphaerosyllis californiensis	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	No barrier

Sphaerosyllis spp.	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	No barrier
Typosyllis nipponica	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Syllidae unidentified	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	No barrier
Eteone lighti	Surface Carnivore	Egg layers with pelagic larvae	No barrier
Eteone sp. A	Surface Carnivore	Egg layers with pelagic larvae	No barrier
Eteone spp.^^	Surface Carnivore	Egg layers with pelagic larvae	No barrier
Phyllodoce longipes	Surface Carnivore	Egg layers with pelagic larvae	No barrier
Podarkiopsis glabrus	Surface Carnivore	Broadcast spawn with pelagic larvae	No barrier
Sigambra ?setosa	Surface Carnivore	Broadcast spawn with pelagic larvae	No barrier
Neanthes spp. (not N. succinea)	Subsurface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Nephtys caecoides	Subsurface Deposit Feeder	Egg layers with pelagic larvae	No barrier
Nephtys cornuta	Subsurface Deposit Feeder	Egg layers with pelagic larvae	No barrier
Nephtys spp.^^^	Subsurface Deposit Feeder	Egg layers with pelagic larvae	No barrier
Glycinde picta (= G. polygnatha)	Subsurface Carnivore	Broadcast spawn with pelagic larvae	No barrier
Glycinde spp.	Subsurface Carnivore	Broadcast spawn with pelagic larvae	No barrier
Glycera robusta	Subsurface Carnivore	Broadcast spawn with pelagic larvae	No barrier
Onuphis spp.	Subsurface Carnivore	Egg layers with pelagic larvae	No barrier
Dorvillea longicornis (= D. rudolphi)	Subsurface Carnivore	Broadcast spawn with pelagic larvae	No barrier
Dorvillea spp.^^^	Subsurface Carnivore	Broadcast spawn with pelagic larvae	No barrier
Pettiboneia gracilis	Subsurface Carnivore	Broadcast spawn with pelagic larvae	No barrier
Lumbrineridae unidentified	Subsurface Deposit Feeder	Brood in tube with crawl away larvae	No barrier
Leitoscoloplos pugettensis	Subsurface Deposit Feeder	Egg layers with pelagic larvae	No barrier
Orbiniidae unidentified	Subsurface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Dipolydora caulleryi	Surface Deposit Feeder	Egg layers with pelagic larvae	Tubed with tissue
Dipolydora socialis	Surface Deposit Feeder	Egg layers with pelagic larvae	Tubed with tissue
Polydora cornuta	Filter & Surface Deposit Feeder	Egg layers with pelagic larvae	Tubed with tissue
Pseudopolydora kempi	Filter & Surface Deposit Feeder	Egg layers with pelagic larvae	Tubed with tissue
Pseudopolydora paucibranchiata	Filter & Surface Deposit Feeder	Egg layers with pelagic larvae	Tubed with tissue
Scolelepis spp.	Filter & Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Spiophanes berkeleyorum	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Spiophanes bombyx	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Spiophanes spp.	Surface Deposit Feeder	Broadcast spawn or brooding possible	Tubed with tissue
Streblospio benedicti	Surface Deposit Feeder	Egg layer to the surface and asexual	Tubed with tissue
Spionidae unidentified^^^	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue

Aricidea spp.	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Chaetozone spp.	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Cirriformia nr. moorei (=sprirabrancha)	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Cirriformia spp.	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Cirratulidae unidentified	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Cossura spp.	Subsurface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Acrocirridae unidentified	Subsurface Deposit Feeder	Broadcast spawn with pelagic larvae	No barrier
Armandia brevis	Subsurface Deposit Feeder	Egg layer to the surface and asexual	No barrier
Capitella capitata complex	Subsurface Deposit Feeder	Broadcast spawn or brooding possible	No barrier
Heteromastus filiformis	Subsurface Deposit Feeder	Broadcast spawn or brooding possible	No barrier
Mediomastus ambiseta/californiensis	Subsurface Deposit Feeder	Egg layer to the surface and asexual	No barrier
Capitellidae unid.	Subsurface Deposit Feeder	Broadcast spawn or brooding possible	No barrier
Sabaco elongatus	Subsurface Deposit Feeder	Broadcast spawn or brooding possible	Tubed with tissue
Maldanidae unidentified	Subsurface Deposit Feeder	Brood in tube with crawl away larvae	Tubed with tissue
Euchone limnicola	Filter Feeder	Brood in tube with crawl away larvae	Tubed with tissue
Sabellidae unidentified	Filter Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Ampharete labrops	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Ampharete spp.	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Ameana occidentalis	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Ameana sp. SF 1	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Tubed with tissue
Ameana spp.****	Surface Deposit Feeder	Broadcast spawn or brooding possible	Tubed with tissue
Pista spp.	Surface Deposit Feeder	Broadcast spawn or brooding possible	Tubed with tissue
Polycirrus californicus	Surface Deposit Feeder	Broadcast spawn or brooding possible	Tubed with tissue
Terebellidae unidentified	Surface Deposit Feeder	Broadcast spawn or brooding possible	Tubed with tissue
Phylum Arthropoda			
Crustacea			
Cephalocarida	Filter Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Lightiella serendipita?	Filter Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Ostracoda	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Cyprideis sp. A	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Eusarsiella zosteracola	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living

Ostracoda A	Surface Deposit Feeder		Chitin barrier and free living
Copepoda	Surface Deposit Feeder	Egg layers with pelagic larvae	Chitin barrier and free living
Calinoida	Surface Deposit Feeder	Egg layers with pelagic larvae	Chitin barrier and free living
Harpacticoida	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Chitin barrier and free living
Cirripedia	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Balanus spp.	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Balanidae unidentified	Filter Feeder		Shell barrier
Leptostraca	Filter Feeder	Ovoviparous/Viviparous direct juvenile release	Shell barrier
Nebaliidae - unidentified	Filter Feeder	Ovoviparous/Viviparous direct juvenile release	Shell barrier
Mycidacea unidentified	Filter Feeder		Chitin barrier and free living
Cumacea	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Cumella vulgaris	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Eudorella pacifica	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Nippoleucon hinnumensis	Surface Deposit Feeder		Chitin barrier and free living
Isopoda	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Munna spp.	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Paranthura japonica	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Synidotea laevidorsalis	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Synidotea spp.	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Tanaidacea	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Leptochelia spp. (replaces L. dubia)	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Amphipoda			Chitin barrier and free living

Americorophium stimpsoni	Filter & Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Ampelisca abdita	Filter Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Ampithoe valida	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Ampithoe spp.	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Corophium heteroceratum	Filter & Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Corophium spp.	Filter & Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Grandiderella japonica	Filter & Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Melita spp.	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	
Monocorophium acherusicum	Filter & Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Monocorophium insidiosum	Filter & Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Monocorophium spp.	Filter & Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Pacificulodes spinnipes	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	
Oedicerodidae unidentified	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Photis brevipes*	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Photis spp.**	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Tubed with chitin
Caprella scaura	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Caprella ?verucosa	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Caprella sp. A female	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Caprella spp. (too small to id to species)	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Tritella pilimana	Surface Deposit Feeder	Ovoviparous/Viviparous direct juvenile release	Chitin barrier and free living
Unidentified Caprellidae	Surface Deposit Feeder		Chitin barrier and free living
Decapoda		Egg layers with pelagic larvae	Chitin barrier and free living
Cancer jordani	Surface Carnivore	Egg layers with pelagic larvae	Chitin barrier and free living
Crangon nigricauda	Surface Carnivore	Egg layers with pelagic larvae	Chitin barrier and free living
Crangon spp.	Surface Carnivore	Egg layers with pelagic larvae	Chitin barrier and free living
Upogebia pugettensis	Filter Feeder	Egg layers with pelagic larvae	Tubed with chitin

Callianassidae unidentified	Filter Feeder	Egg layers with pelagic larvae	Tubed with chitin
Paguridae (megalopa)	Filter Feeder	Egg layers with pelagic larvae	Chitin barrier and free living
Pinnotheridae	Filter Feeder		Chitin barrier and free living
Pycnogonida	Surface Carnivore	Egg layer with surface juveniles	Chitin barrier and free living
Achelia chelata?	Surface Carnivore		Chitin barrier and free living
Mollusca			
Gastropoda			
Crepidula convexa	Filter Feeder	Viviparous/oviparous with surface juvenile	Shell barrier
Crepidula plana	Filter Feeder	Viviparous/oviparous with surface juvenile	Shell barrier
Crepidula spp.	Filter Feeder	Viviparous/oviparous with surface juvenile	Shell barrier
Haminoea japonica	Surface Deposit Feeder	Egg layer with surface juveniles	No barrier
Haminoea spp.	Surface Deposit Feeder	Egg layer with surface juveniles	No barrier
Philine spp. ***	Surface Carnivore	Egg layer with surface juveniles	No barrier
Unidentified Nudibranchia	Surface Carnivore	Egg layer with surface juveniles	Shell barrier
Unidentified Gastropoda^		Viviparous/oviparous with surface juvenile	Shell barrier
Pelycepoda			Shell barrier
Corbula amurensis	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Cryptomya californica	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Gemma gemma	Filter Feeder	Ovoviparous/Viviparous direct juvenile release	Shell barrier
Lyonsia californica	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Macoma balthica/petalum	Filter & Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Macoma spp.	Filter & Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Modiolus rectus	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Musculista senhousia	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Mactridae unidentified^	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Mya arenaria	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Mytilus spp.	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Nuculana spp.	Subsurface Deposit Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Rochefortia coani	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Rochefortia tumida	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Siliqua lucida	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier

Theora lubrica^	Surface Deposit Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Venerupis philippinarum	Filter Feeder	Broadcast spawn with pelagic larvae	Shell barrier
Unidentified Bivalvia^		Broadcast spawn with pelagic larvae	Shell barrier
Bryozoa	Filter Feeder		Shell barrier
Chordata	Filter Feeder		
Clavelina spp.	Filter Feeder	Broadcast spawn or fission with pelagic larvae	No barrier
Molgula spp.	Filter Feeder	Broadcast spawn with pelagic larvae	No barrier
Metandrocarpa spp. ?	Filter Feeder	Egg layer to the surface and asexual	No barrier

3. Information about the chemical data stations

Sample identification, location, and depth

Site Code	Cruise #	Sample Date	Latitude	Longitude	depth [m]
BB30	1993-03	03/11/1993	37.67016	-122.3295	9.00
BB30	1993-09	09/22/1993	37.66933	-122.32933	9.00
BB30	1994-02	02/15/1994	37.6705	-122.32966	9.00
BB30	1994-08	08/29/1994	37.66933	-122.32933	8.00
BB30	1995-02	02/20/1995	37.66983	-122.32933	9.00
BB30	1995-08	08/30/1995	37.6695	-122.32883	9.00
BB30	1996-02	02/21/1996	37.67016	-122.329	8.00
BB30	1996-07	08/01/1996	37.6695	-122.32916	10.00
BB30	1997-01	02/05/1997	37.67055	-122.33666	10.00
BB30	1997-08	08/12/1997	37.67016	-122.3295	9.00
BB30	1998-02	02/10/1998	37.67016	-122.3295	9.00
BB30	1998-07	08/04/1998	37.67016	-122.3295	9.00
BB30	1999-02	02/17/1999	37.67016	-122.3295	9.00
BB30	1999-07	07/27/1999	37.66866	-122.32883	9.00
BB30	2000-07	07/25/2000	37.66866	-122.32883	9.00
BB30	2001-08	08/14/2001	37.66866	-122.32883	8.00
BB70	1994-02	02/15/1994	37.74725	-122.3235	10.00
BB70	1994-08	08/30/1994	37.7475	-122.32316	10.00
BB70	1995-02	02/21/1995	37.74683	-122.3235	10.00
BB70	1995-08	08/28/1995	37.74666	-122.323	10.00
BB70	1996-02	02/21/1996	37.74683	-122.32333	10.00
BB70	1996-07	08/01/1996	37.747	-122.32283	11.00
BB70	1997-01	02/03/1997	37.75527	-122.32694	9.00
BB70	1997-08	08/12/1997	37.74733	-122.32333	10.00
BB70	1998-02	02/10/1998	37.74733	-122.32333	10.00
BB70	1998-07	08/03/1998	37.74733	-122.32333	10.00
BB70	1999-02	02/17/1999	37.74733	-122.32333	10.00
BB70	1999-07	07/26/1999	37.74666	-122.32233	10.50
BB70	2000-07	07/24/2000	37.74666	-122.32233	10.50
BB70	2001-08	08/14/2001	37.7465	-122.32216	9.00
BC41	1993-03	03/11/1993	37.889	-122.3425	1.50
BC41	1993-09	09/21/1993	37.88933	-122.342	1.50
BC41	1994-02	02/14/1994	37.8885	-122.34216	2.00
BC41	1994-08	08/29/1994	37.88833	-122.34233	3.00
BC41	1995-02	02/20/1995	37.8885	-122.343	3.00
BC41	1995-08	08/28/1995	37.88966	-122.34233	2.00
BC41	1996-02	02/16/1996	37.88916	-122.34183	1.00
BC41	1996-07	08/02/1996	37.88916	-122.34216	2.00
BC41	1997-01	02/03/1997	37.89277	-122.34861	3.00
BC41	1997-08	08/11/1997	37.889	-122.3425	1.50
BC41	1998-02	02/09/1998	37.889	-122.3425	1.50
BC41	1998-07	08/03/1998	37.889	-122.3425	1.50
BC41	1999-02	02/16/1999	37.889	-122.3425	1.50
BC41	1999-07	07/26/1999	37.88783	-122.3425	2.00
BC41	2000-07	07/24/2000	37.88783	-122.3425	2.00
BC41	2001-08	08/13/2001	37.8875	-122.34283	2.00
CB001S	2004-07	07/29/2004	37.87587	-122.3621	3.80
CB001S	2005-08	08/26/2005	37.87613	-122.36172	3.60
CB001S	2006-08	08/04/2006	37.87527	-122.36205	3.70
CB001S	2007-08	08/24/2007	37.87672	-122.36163	3.40
CB004S	2007-08	08/23/2007	37.75495	-122.33032	11.10
CB005S	2002-07	08/08/2002	37.853	-122.325	2.00
CB016S	2004-07	07/30/2004	37.696	-122.36542	7.70
CB020S	2004-07	07/30/2004	37.749	-122.30892	6.50
CB025S	2005-08	08/25/2005	37.86668	-122.3784	5.60
CB028S	2006-08	08/07/2006	37.70992	-122.36415	3.70
CB029S	2006-08	08/04/2006	37.84675	-122.35205	4.80
CB030S	2006-08	08/07/2006	37.75817	-122.27487	2.30
CB032S	2006-08	08/07/2006	37.67643	-122.35355	8.50
CB033S	2007-08	08/24/2007	37.86732	-122.36347	3.60
CB080S	2005-08	08/25/2005	37.71623	-122.34937	15.60
PC-16	HP VS		37.9017	-122.4595	
SB-16	HP VS		37.7223	-122.3763	0.00
SB-17	HP VS		37.7216	-122.3762	0.00
SB-11	HP VS		37.7235	-122.3775	0.00
SB-20	HP VS		37.7234	-122.377	0.00
SB-22	HP VS		37.7230	-122.3766	0.00

Candlestick	CB004S, CB016S, CB028S, BB70
Alameda	CB030S, CB020S, CB032S, CB080S, BB30
Emeryville	CB005S, CB029S
Richmond	BC41, CB033S, CB001S
Tiburon	PC-16
Hunters Point	SB-16, SB-17, SB-11, SB-20, SB-22

Physical properties and TOC content

Site Code	Cruise #	Clay	Fine	Granule	Sand	Silt	TOC
		<0.0039 mm	<0.0625 mm	and Peddle	0.0625 to	0.0039 to	
				2.0 to <64 mm	<2.0 mm	<0.0625 mm	
		%	%	%	%	%	%
BB30	1993-03	50.	73.	0.	27.	23.	1.40
BB30	1993-09	43.	61.	12.	27.	18.	1.00
BB30	1994-02	29.	43.	3.	53.	14.	0.72
BB30	1994-08	41.	61.	2.	36.	20.	1.17
BB30	1995-02	46.	67.	6.	27.	21.	1.17
BB30	1995-08	48.	68.	2.	30.	20.	1.05
BB30	1996-02	49.	77.	3.	20.	28.	1.02
BB30	1996-07	57.	85.	0.	14.	28.	1.29
BB30	1997-01	49.	77.	2.	21.	29.	1.29
BB30	1997-08	41.	59.	2.	39.	18.	0.80
BB30	1998-02	49.	75.	0.	25.	25.	1.12
BB30	1998-07	35.	50.	10.	40.	15.	0.93
BB30	1999-02	36.	57.	5.	38.	21.	1.31
BB30	1999-07	29.	45.	7.	48.	16.	0.87
BB30	2000-07	38.	56.	3.	41.	18.	0.98
BB30	2001-08	41.	62.	1.	36.	22.	1.00
BB70	1994-02	56.	82.		18.	26.	1.06
BB70	1994-08	44.	70.		30.	25.	0.95
BB70	1995-02	50.	76.	0.	24.	26.	1.01
BB70	1995-08	69.	97.	0.	5.	28.	2.22
BB70	1996-02	51.	80.	0.	20.	29.	1.03
BB70	1996-07	53.	85.	0.	15.	32.	1.19
BB70	1997-01	59.	90.	0.	10.	31.	1.19
BB70	1997-08	45.	64.	0.	36.	20.	1.06
BB70	1998-02	46.	74.	0.	26.	28.	1.06
BB70	1998-07	50.	70.	0.	30.	21.	0.91
BB70	1999-02	42.	72.	0.	28.	30.	1.09
BB70	1999-07	46.	69.	0.	31.	23.	0.93
BB70	2000-07	47.	70.	0.	30.	23.	1.05
BB70	2001-08	43.	65.	0.	35.	22.	0.92
BC41	1993-03	50.	87.	0.	14.	37.	1.00
BC41	1993-09	53.	86.	0.	14.	33.	1.00
BC41	1994-02	50.	84.	0.	16.	34.	1.01
BC41	1994-08	46.	87.	0.	13.	41.	0.94
BC41	1995-02	54.	89.	0.	11.	35.	1.04
BC41	1995-08	51.	88.	0.	12.	37.	1.12
BC41	1996-02	47.	87.	0.	13.	40.	1.00
BC41	1996-07	56.	90.	0.	10.	34.	1.20
BC41	1997-01	49.	89.	0.	11.	40.	1.20
BC41	1997-08	53.	92.	0.	8.	39.	1.06
BC41	1998-02	59.	91.		9.	32.	1.06
BC41	1998-07	50.	82.		18.	33.	1.20
BC41	1999-02	53.	84.	1.	15.	31.	1.06
BC41	1999-07	57.	87.	0.	13.	30.	1.08
BC41	2000-07	50.	83.	0.	18.	33.	1.11
BC41	2001-08	45.	74.	0.	26.	29.	0.97
CB001S	2004-07	40.	67.	0.	33.	27.	0.89
CB001S	2005-08	43.	83.	0.	17.	40.	1.83
CB001S	2006-08	31.	55.	0.	45.	24.	1.09
CB002S	2002-07	63.	93.	0.	7.	30.	1.18
CB004S	2007-08	11.	21.	1.	78.	10.	0.68
CB005S	2002-07	57.	90.	0.	10.	34.	1.07
CB016S	2004-07	48.	80.	0.	20.	32.	1.17
CB020S	2004-07	52.	86.	0.	14.	34.	1.10
CB025S	2005-08	31.	72.	0.	28.	41.	0.85
CB028S	2006-08	58.	91.	0.	9.	33.	1.28
CB029S	2006-08	32.	57.	0.	43.	25.	0.81
CB030S	2006-08	14.	22.	8.	70.	8.	0.40
CB032S	2006-08	61.	93.	0.	7.	32.	1.29
CB033S	2007-08	42.	87.		13.	45.	0.54
CB080S	2005-08	34.	55.	0.	44.	22.	0.82
PC-16	HP VS						0.995
SB-16	HP VS		47.4				1.36
SB-17	HP VS						
SB-11	HP VS		95.1				1.63
SB-20	HP VS		92.7				1.7
SB-22	HP VS		87.4				1.7
Cho 2007							1.1
Cho 2007							0.7

Sediment chemistry for the selected stations in the Central Bay
available from the SFEI data base

		Total PCBs	total PAHs	total LPAHs	total DDT	Dieldrin		As	Cu	Hg	Ni	Pb
Site Code	Cruise #	µg/Kg	µg/Kg	µg/Kg	µg/Kg	µg/Kg		mg/Kg	mg/Kg	mg/Kg	mg/Kg	mg/Kg
San Francisco Estuarine Institute online data base (as of August 2010)												
BB30	1993-03	11.56	1,715.7	153.0	1.95	0.06		14.15	28.90	0.27	61.20	21.60
BB30	1993-09	3.36	2,154.9	248.6	1.54			10.20	29.44	0.25	59.44	15.52
BB30	1994-02	17.62	4,022.6	526.1	2.72	0.51		9.70	38.42	0.27	70.31	14.42
BB30	1994-08	9.40	1,407.6	158.2	1.97			8.67	33.15	0.24	86.35	16.40
BB30	1995-02	4.93	2,637.0	344.0	0.88			6.90	35.69	0.28	69.31	20.40
BB30	1995-08	10.75	1,363.0	181.0	2.79			9.38	34.90	0.27	76.40	26.40
BB30	1996-02	10.05	3,290.0	502.0	1.60	0.10		9.06	46.24	0.24	101.06	16.84
BB30	1996-07	14.81	2,324.0	277.0	3.70	0.17		9.98	38.80	0.26	85.60	21.40
BB30	1997-01	6.24	3,125.0	529.0	3.40			5.53	34.30	0.23	87.40	18.00
BB30	1997-08	4.01	808.0	107.0	1.10			7.87	29.80	0.14	85.20	14.00
BB30	1998-02	9.95	2,642.0	398.0	6.90			8.27	37.40	0.26	82.00	14.00
BB30	1998-07	7.77	1,440.0	165.0	1.10			8.40	28.00	0.18	85.30	16.00
BB30	1999-02	10.38	2,263.0	299.0	6.00				32.05	0.14	71.24	14.60
BB30	1999-07	8.28	1,579.0	237.0	3.00			6.90	24.45	0.15	54.55	13.39
BB30	2000-07		1,349.4	212.0				5.52	30.00	0.13	68.57	16.10
BB30	2001-08	9.92	1,731.8	167.8	2.13			8.55	32.17	0.20	52.65	15.07
BB70	1994-02	16.83	6,085.1	834.8	4.60	0.26		12.03	43.94	0.31	63.98	19.22
BB70	1994-08	30.74	2,327.7	345.0	3.51			9.92	39.06	0.33	89.99	25.10
BB70	1995-02	11.56	3,720.0	938.0	1.26			9.80	41.42	0.33	78.67	37.77
BB70	1995-08	3.88	2,279.0	475.0	4.63			13.38	42.10	0.32	80.20	22.10
BB70	1996-02	11.22	1,414.0	208.0	2.70	0.15		9.59	48.06	0.28	98.19	17.53
BB70	1996-07	8.01	2,169.0	306.0	1.60			12.60	36.00	0.28	83.10	20.80
BB70	1997-01	1.14	1,329.0	167.0	2.80			8.05	51.60	0.27	116.00	13.00
BB70	1997-08	27.83	3,337.0	586.0	4.30			8.07	38.70	0.24	93.40	22.60
BB70	1998-02	14.22	2,550.0	275.0	7.50			12.00	40.60	0.23	103.00	17.00
BB70	1998-07	14.40	3,476.0	431.0	4.00			9.20	40.10	0.30	104.00	29.00
BB70	1999-02	8.86	3,879.0	522.0	9.10			9.10	37.34	0.20	72.96	17.96
BB70	1999-07	22.50	3,194.0	467.0	30.20			8.00	38.63	0.31	68.62	24.30
BB70	2000-07	26.58	4,843.8	744.7	4.23			7.57	46.01	0.41	79.59	28.10
BB70	2001-08	22.31	2,970.6	438.8	2.88			7.84	46.45	0.42	70.26	27.95
BC41	1993-03	20.41	2,169.7	215.4	7.02	0.15		29.41	37.70	0.30	71.90	29.70
BC41	1993-09	39.70	1,867.4	170.9	2.28			12.40	35.12	0.33	72.82	23.41
BC41	1994-02	9.41	5,420.0	655.8	7.15	0.34		8.18	43.67	0.31	79.29	19.26
BC41	1994-08	8.91	1,609.5	203.8	2.97			12.30	37.80	0.27	82.99	20.40
BC41	1995-02	5.26	2,955.0	427.0	1.34			11.99	40.08	0.33	73.13	24.03
BC41	1995-08	8.51	2,206.0	343.0	4.03	0.29		11.86	36.40	0.32	81.30	28.80
BC41	1996-02	8.08	1,809.0	257.0	3.10	0.16		14.70	46.94	0.27	91.97	21.13
BC41	1996-07	10.14	2,395.0	304.0	5.00	0.20		12.80	39.10	0.28	86.60	22.90
BC41	1997-01	13.98	1,682.0	203.0	9.40			6.73	45.00	0.28	90.70	17.00
BC41	1997-08	11.06	2,369.0	294.0	6.00			10.40	41.70	0.25	98.90	23.40
BC41	1998-02	12.13	1,516.0	181.0	10.30			13.37	50.10	0.29	123.00	22.00
BC41	1998-07	3.11	2,046.0	273.0	2.10			11.00	40.30	0.25	100.00	24.00
BC41	1999-02	7.2	2,554.0	332.0	7.90			10.00	39.47	0.24	78.42	20.09
BC41	1999-07	3.0	1,410.0	186.0	2.30			9.90	42.44	0.24	77.68	20.26
BC41	2000-07	5.6	2,377.1	380.0	5.27			8.43	44.14	0.15	83.00	24.99
BC41	2001-08	12.8	3,097.4	323.4	7.00			9.68	35.63	0.36	63.13	19.19
CB001S	2004-07	5.9	3,152.1	439.1	2.22	0.07		8.37	41.43	0.33	84.60	23.68
CB001S	2005-08	6.7	2,861.1	481.3	2.98	0.11		9.86	37.69	0.22	69.84	18.13
CB001S	2006-08	3.5	3,094.2	330.7	9.54	0.08		7.04	42.47	0.27	83.39	20.40
CB001S	2007-08	13.8	2,966.2	390.9	3.89	0.12		9.10	34.29	0.30	71.17	20.23
CB002S	2002-07		3,536.0	356.6	0.97			10.23	42.35	0.30	82.25	22.22
CB004S	2007-08	4.6	429.5	37.9	0.99	0.03		4.70	12.77	0.34	35.31	9.81
CB005S	2002-07		2,578.1	283.5	0.39			6.53	19.30	0.26	36.59	11.88
CB016S	2004-07	8.7	5,257.8	664.3	2.01	0.07		7.25	48.80	0.27	80.83	22.38
CB020S	2004-07	2.4	3,148.3	383.7	0.39	0.02		6.32	47.12	0.30	93.55	26.69
CB025S	2005-08	5.8	5,813.7	854.6	1.83	0.06		9.22	29.24	0.17	62.83	14.39
CB028S	2006-08	5.3	11,478.4	1,406.4	1.20	0.06		7.80	38.57	0.25	76.18	18.68
CB029S	2006-08	6.3	3,228.5	411.4	1.74	0.09		6.77	28.30	0.19	66.48	16.75
CB030S	2006-08	1.5	540.3	55.1	0.30	0.02		2.60	12.75	0.10	25.52	7.99
CB032S	2006-08	7.2	3,896.6	286.0	1.16	0.05		7.76	46.79	0.28	90.61	22.25
CB033S	2007-08	10.2	3,021.6	367.9	4.96	0.27		10.10	36.33	0.24	72.62	18.55
CB080S	2005-08	5.1	12,210.3	2,681.7	0.88	0.04		7.83	25.36	0.15	66.09	14.43
Hunters Point Validation Study												
PC-16	HP VS	3.3	867.6	91	5.5			11.5	40.1	0.289	86.9	21.6
SB-16	HP VS	1786.0	4148	276	5.09	0.08		11.3	173	0.79	199	139
SB-11	HP VS	1113.0	2538	248	17							
SB-20	HP VS	1565.0	2974	281	16	3.63			133	1.06	118	106
SB-22	HP VS	1817.0	2931	234	10	2.75			163	1.26	130	142

Additional stations for TOC and total PCB concentrations in sediment
provided by the SFEI but not available from the online data base

StudyID	StationID	Latitude	Longitude	SampleID	TOC [%]	Total PCBs [µg/Kg]
ANAS_SEAPL	RL-5	37.89938	-122.46478	124-RL5-011	1.09	7.082
BPTCP	1219	37.898833	-122.464333	20005_Rep1	1.13	10.732
BPTCP	1220	37.899167	-122.464167	20005_Rep2	1.09	8.700
BPTCP	1221	37.900167	-122.465167	20005_Rep3	1.13	8.231
NOAA_01	20-2	37.89798	-122.33545	1	0.93	9.680
NOAA_01	20-3	37.87605	-122.36068	1	0.99	6.889
NOAA_01	27-3	37.71192	-122.37233	1	1.37	13.004
NOAA_01	30-2	37.75888	-122.28347	1	0.89	5.070
WEMAP00	20-5	37.85431	-122.33817	20-5	0.92	6.300
WEMAP00	35-2	37.68912	-122.35265	35-2	1.27	13.307
WEMAP00	CA00-0032	37.904752	-122.465829	32	1.47	6.890
WEMAP00	CA00-0034	37.878739	-122.387019	34	1.16	
WEMAP00	CA00-0049	37.674707	-122.368832	49	37.50	37.500
Candlestick	NOAA_01 (27-3), WEMAP00 (32-2), WEMAP00 (CA00-0049)					
Alameda	NOAA_01 (30-2)					
Emeryville	WEMAP00 (20-5)					
Richmond	NOAA_01 (20-2), NOAA_01 (20-3), WEMAP00 (CA00-0034)					
Tiburon	ANAS_SEAPL (RL-5), BPTCP (1219), BPTCP (1220), BPTCP (1221), WEMAP00 (CA00-032)					

4. Total abundance data

April 2007 survey

DATE: April, 2007							
	STATION						
TAXON	CB 1	CB 2	CB 4	CB 5	CB 8	CB 9	CB 12
Cnidaria							
Hydrozoa	+						
Anthozoa							
Actiniaria					6		
Burrowing anenome (#1)						1	
Edwardsiidae (?Scolanthus spp)				1			
Attached (#2)						1	5
Unid. Actiniaria (Diadumene?)							
Alcyonacea - Pennatulidae							
Stylatula spp.							2
Turbellaria							
Nemertea		4					
Tubulanus spp.			7	3			1
Unidentified Nemertea						3	
Nematoda	5				24	3	
Phoronida							
Phoronis spp.							
Annelida							
Oligochaeta	57		1				
Naididae							
Tubificidae				1			
Tubificoides wasselli							
Tubificoides spp.				1			
Polychaeta							
Tenonia priops							
Harmothoe imbricata	4	1		3	5	1	2
Malmgreniella macginintiei							
Malmgreniella spp. (= Harmothoe spp.)		1	2	1			
Exogone lourei	1	2		2		2	
Exogone spp.			1				
Sphaerosyllis californiensis		2	12	3	4	2	4
Sphaerosyllis spp.							
Typosyllis nipponica	1	1	1				
Syllidae unidentified							
Eteone sp. A							
Phyllodoce longipes	9	1		2			
Podarkiopsis glabrus							
Sigambra ?setosa							
Nephtys spp.^^^				1			
Nephtys caecoides	1	1					
Nephtys cornuta	4	4	22			2	
Glycinde picta (= G. polygnatha)	4	7	7	7	2	5	4
Glycinde spp.				1			1
Dorvillea longicornis (= D. rudolphi)	2	1	1	4		11	
Dorvillea spp.^^^							

DATE: April, 2007							
	STATION						
TAXON	CB 1	CB 2	CB 4	CB 5	CB 8	CB 9	CB 12
Pettiboneia gracilis			3				
Lumbrineridae unidentified							
Leitoscoloplos pugettensis							
Orbiniidae unidentified							
Dipolydora caulleryi							
Polydora cornuta							
Pseudopolydora kempi		1					
Pseudopolydora paucibranchiata	6				6		1
Scolelepis spp.							
Spiophanes bombyx							
Spiophanes spp.							1
Streblospio benedicti							
Aricidea spp.							
Chaetozone spp.	1						
Cirriformia nr. moorei (=sprirabrancha)			1				
Cirratulidae unidentified			1				
Cossura spp.							
Acrocirridae unidentified			1				
Armandia brevis					1		
Capitella capitata complex	3				4		
Heteromastus filiformis	1						
Mediomastus spp.				2			
Mediomastus ambiseta/californiensis	4	3	8		1	7	
Notomastus tenuis				1			
Capitellidae unid.							
Sabaco elongatus	2			4	42	2	136
Maldanidae unidentified	1						
Euchone limnicola	3	6	5	7	50	1	18
Ampharete labrops				1			
Ameana occidentalis							
Ameana sp. SF 1	2	7	6				
Ameana spp.****							
Pista spp.							
Polycirrus californicus							
Terebellidae unidentified							
Phylum Arthropoda							
Crustacea							
Ostracoda							
Eusarsiella zosteracola							
Copepoda							
Calinoida							
Harpacticoida							
Cirripedia							
Balanidae unidentified							3

DATE: April, 2007							
	STATION						
TAXON	CB 1	CB 2	CB 4	CB 5	CB 8	CB 9	CB 12
Mycidacea unidentified							
Cumacea							
Cumella vulgaris			2				
Eudorella pacifica	4		28				
Nippoleucon hinnumensis	90	130	14	4	43	2	35
Isopoda							
Munna spp.							
Paranthura japonica							
Synidotea laevidorsalis							
Unid. Sphaeromatidae							
Tanaidacea							
Leptochelia spp. (replaces L. dubia)	2	3	2	102	25	84	1
Amphipoda							
Ampelisca abdita	481	368	47	48	315	59	222
Corophium heteroceratum	1	105	403	18	2	22	17
Corophium spp.							
Grandiderella japonica					3	1	
Monocorophium acherusicum	84	68	10		206		36
Monocorophium insidiosum					7		
Monocorophium spp.	4		8	**1**	3		60
Oedicerodidae unidentified							
Photis brevipes*	7	4	8				
Photis spp.**	31	7				1	
Caprella scaura		2			8		
Caprella ?verucosa							
Caprella sp. A female					3		
Caprella spp. (too small to id to species)	2	1	1	**3**	48		1
Unidentified Caprellidae				**5**		1	
Stenothoe valida				**1**			
Decapoda							
Crangon nigricauda	1						
Upogebia pugettensis							1
Callianassidae unidentified						2	
Paguridae (megalopa)							
Pycnogonida							
Ammotheidae							
Achelia chelata?							
Leptopcarida						1	
Mollusca							
Gastropoda							
Crepidula convexa							
Haminoea japonica							
Philine spp. ***				1	2		
Unidentified Nudibranchia					1		

DATE: April, 2007							
	STATION						
TAXON	**CB 1**	**CB 2**	**CB 4**	**CB 5**	**CB 8**	**CB 9**	**CB 12**
Unidentified Gastropoda^							
Pelycepoda							
Potamocorbula amurensis					1		
Cryptomya californica							
Gemma gemma							1
Lyonsia californica						1	
Macoma balthica/petalum							
Macoma spp.							
Modiolus rectus		1					
Musculista senhousia	4	1			1		6
Mytilus spp.	1			1			
Rochefortia tumida			3				
Siliqua lucida	1						
Theora lubrica	2	2	2				
Venerupis philippinarum							4
Unidentified Bivalvia^				1			
Bryozoa							+
Chordata							
Molgula spp.					1		1
Metandrocarpa spp. ?							
Total	**826**	**734**	**607**	**230**	**814**	**215**	**563**

DATE: April, 2007							
	STATION						
TAXON	CB 15	CB 16	CB 17	CB 20	CB 22	CB 24	CB 25
Cnidaria							
Hydrozoa			+				
Anthozoa							
Actiniaria	4						
Burrowing anenome (#1)		1	3				
Edwardsiidae (?Scolanthus spp)							
Attached (#2)			2				14
Unid. Actiniaria (Diadumene?)						23	
Alcyonacea - Pennatulidae							
Stylatula spp.	1					1	
Turbellaria	1		2	1	1		
Nemertea							
Tubulanus spp.		1			2		
Unidentified Nemertea			4	1	3	2	
Nematoda	9		2		5		24
Phoronida							
Phoronis spp.					2		
Annelida							
Oligochaeta					2		4
Naididae						1	
Tubificidae						2	
Tubificoides wasselli						101	
Tubificoides spp.							
Polychaeta							
Tenonia priops					1		
Harmothoe imbricata		3					7
Malmgreniella macginintiei					7		
Malmgreniella spp. (= Harmothoe spp.)							
Exogone lourei			7			202	28
Exogone spp.							
Sphaerosyllis californiensis	27	4	17	18	21	38	53
Sphaerosyllis spp.			1			1	
Typosyllis nipponica	6		2				
Syllidae unidentified							
Eteone sp. A					2	1	
Phyllodoce longipes	4				6	1	
Podarkiopsis glabrus					1		
Sigambra ?setosa					2		
Nephtys spp.^^^							
Nephtys caecoides							
Nephtys cornuta					38		
Glycinde picta (= G. polygnatha)	3	7	1	2	16	11	6
Glycinde spp.						1	3
Dorvillea longicornis (= D. rudolphi)	3		5	5	4	48	
Dorvillea spp.^^^							7

DATE: April, 2007							
	STATION						
TAXON	CB 15	CB 16	CB 17	CB 20	CB 22	CB 24	CB 25
Pettiboneia gracilis							
Lumbrineridae unidentified							
Leitoscoloplos pugettensis						1	1
Orbiniidae unidentified							
Dipolydora caulleryi							
Polydora cornuta				1			
Pseudopolydora kempi						2	
Pseudopolydora paucibranchiata	5	3		2		20	
Scolelepis spp.						1	
Spiophanes bombyx			3				
Spiophanes spp.							
Streblospio benedicti							
Aricidea spp.			1				
Chaetozone spp.					4	20	
Cirriformia nr. moorei (=sprirabrancha)						1	
Cirratulidae unidentified				1			
Cossura spp.			3			2	
Acrocirridae unidentified							
Armandia brevis						2	
Capitella capitata complex		2	3	3	3		
Heteromastus filiformis		2		3		5	
Mediomastus spp.							
Mediomastus ambiseta/californiensis	11		7		75		
Notomastus tenuis							
Capitellidae unid.						3	
Sabaco elongatus	28	151	7	81			
Maldanidae unidentified							
Euchone limnicola	15	28	7	14	3		
Ampharete labrops					2		
Ameana occidentalis							
Ameana sp. SF 1			3				
Ameana spp.****					20		
Pista spp.							
Polycirrus californicus						683	
Terebellidae unidentified							
Phylum Arthropoda							
Crustacea							
Ostracoda							
Eusarsiella zosteracola		1			1		
Copepoda							
Calinoida							
Harpacticoida							
Cirripedia							
Balanidae unidentified							

DATE: April, 2007							
	STATION						
TAXON	CB 15	CB 16	CB 17	CB 20	CB 22	CB 24	CB 25
Mycidacea unidentified							
Cumacea							
Cumella vulgaris					1		
Eudorella pacifica					217		
Nippoleucon hinnumensis	10	85		20	13		
Isopoda							
Munna spp.					5		
Paranthura japonica	1					4	1
Synidotea laevidorsalis						1	
Unid. Sphaeromatidae							
Tanaidacea							
Leptochelia spp. (replaces L. dubia)		2	81	28	202	11	27
Amphipoda							
Ampelisca abdita	730	940	7	146	53	7	1
Corophium heteroceratum	12		6	1	151	1	
Corophium spp.						32	
Grandiderella japonica		2		5			
Monocorophium acherusicum	75	38	4	9		603	6
Monocorophium insidiosum				1			3
Monocorophium spp.		85	2	**17**	2	**8**	**5**
Oedicerodidae unidentified							
Photis brevipes*					7		
Photis spp.**			2		9		
Caprella scaura	33	5					
Caprella ?verucosa	3						
Caprella sp. A female							
Caprella spp. (too small to id to species)	16	6	2	**1**		**3**	
Unidentified Caprellidae					2	**34**	
Stenothoe valida							
Decapoda							
Crangon nigricauda							
Upogebia pugettensis							
Callianassidae unidentified							
Paguridae (megalopa)							
Pycnogonida							
Ammotheidae						**1**	
Achelia chelata?				**1**			
Leptopcarida							
Mollusca							
Gastropoda							
Crepidula convexa							27
Haminoea japonica							13
Philine spp. ***						1	3
Unidentified Nudibranchia							

DATE: April, 2007							
	STATION						
TAXON	**CB 15**	**CB 16**	**CB 17**	**CB 20**	**CB 22**	**CB 24**	**CB 25**
Unidentified Gastropoda^							
Pelycepoda							
Potamocorbula amurensis							
Cryptomya californica							
Gemma gemma					1	1	
Lyonsia californica							
Macoma balthica/petalum							
Macoma spp.					1		
Modiolus rectus					1		
Musculista senhousia		4			6	11	5
Mytilus spp.	1	1	1				
Rochefortia tumida							
Siliqua lucida							
Theora lubrica	2				6		
Venerupis philippinarum						3	
Unidentified Bivalvia^			1			1	
Bryozoa							
Chordata							
Molgula spp.							
Metandrocarpa spp. ?				1			
Total	**1000**	**1371**	**186**	**362**	**898**	**1895**	**238**

DATE: April, 2007							
	STATION						
TAXON	CB 27	CB 28	CB 31	CB 33	CB 35	CB 38	CB 41
Cnidaria							
Hydrozoa							
Anthozoa							
Actiniaria							
Burrowing anenome (#1)			4	1		9	3
Edwardsiidae (?Scolanthus spp)							
Attached (#2)	12	36	5				
Unid. Actiniaria (Diadumene?)							
Alcyonacea - Pennatulidae							
Stylatula spp.							
Turbellaria							
Nemertea							
Tubulanus spp.							
Unidentified Nemertea							
Nematoda	2	32	5				5
Phoronida							
Phoronis spp.						1	
Annelida							
Oligochaeta	9	353	64	3			1
Naididae							
Tubificidae							
Tubificoides wasselli							
Tubificoides spp.							
Polychaeta							
Tenonia priops							
Harmothoe imbricata	1	31	12	1	1	5	5
Malmgreniella macginintiei							
Malmgreniella spp. (= Harmothoe spp.)							
Exogone lourei	6	693	73				
Exogone spp.			2				
Sphaerosyllis californiensis	8	14	24				
Sphaerosyllis spp.							
Typosyllis nipponica		1					
Syllidae unidentified		5					
Eteone sp. A							
Phyllodoce longipes					2		
Podarkiopsis glabrus							
Sigambra ?setosa							
Nephtys spp.^^^							
Nephtys caecoides							
Nephtys cornuta							
Glycinde picta (= G. polygnatha)	8	7	7	5	5	2	
Glycinde spp.	6	4	4	2		1	1
Dorvillea longicornis (= D. rudolphi)	19	88	29	1	1	26	1
Dorvillea spp.^^^							

DATE: April, 2007							
	STATION						
TAXON	CB 27	CB 28	CB 31	CB 33	CB 35	CB 38	CB 41
Pettiboneia gracilis							
Lumbrineridae unidentified							
Leitoscoloplos pugettensis				1	1	9	4
Orbiniidae unidentified							1
Dipolydora caulleryi		1					
Polydora cornuta							
Pseudopolydora kempi				3			3
Pseudopolydora paucibranchiata		7	5	36	4	24	32
Scolelepis spp.							
Spiophanes bombyx							
Spiophanes spp.							
Streblospio benedicti				1			
Aricidea spp.							
Chaetozone spp.							
Cirriformia nr. moorei (=sprirabrancha)			2	4	4	15	
Cirratulidae unidentified							
Cossura spp.							
Acrocirridae unidentified							
Armandia brevis	1		1			1	
Capitella capitata complex		6				2	5
Heteromastus filiformis	3	17	4	8	9	10	10
Mediomastus spp.							
Mediomastus ambiseta/californiensis		2			38		
Notomastus tenuis							
Capitellidae unid.							
Sabaco elongatus	5	2	2	13		7	8
Maldanidae unidentified							
Euchone limnicola				10	7	9	6
Ampharete labrops							
Ameana occidentalis							
Ameana sp. SF 1							2
Ameana spp.****			1		3		
Pista spp.			1				
Polycirrus californicus		245	2	22		8	16
Terebellidae unidentified					2		
Phylum Arthropoda							
Crustacea							
Ostracoda							
Eusarsiella zosteracola							
Copepoda							
Calinoida							
Harpacticoida							
Cirripedia							
Balanidae unidentified							

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	STATION						
TAXON	CB 27	CB 28	CB 31	CB 33	CB 35	CB 38	CB 41
Mycidacea unidentified	1						
Cumacea							
Cumella vulgaris							
Eudorella pacifica							
Nippoleucon hinnumensis	3	3		100	4	19	60
Isopoda							
Munna spp.							
Paranthura japonica		2	1		2		
Synidotea laevidorsalis							
Unid. Sphaeromatidae		1					
Tanaidacea							
Leptochelia spp. (replaces L. dubia)	10		55				
Amphipoda							
Ampelisca abdita	7	15	25	37	23	38	21
Corophium heteroceratum			1	21	21	45	50
Corophium spp.							
Grandiderella japonica							1
Monocorophium acherusicum	9	192	64	1	7	2	3
Monocorophium insidiosum	2	2	2			1	1
Monocorophium spp.	**13**	**2**	190	2	3		5
Oedicerodidae unidentified							1
Photis brevipes*							
Photis spp.**							
Caprella scaura							
Caprella ?verucosa							
Caprella sp. A female							
Caprella spp. (too small to id to species)	**2**				1		
Unidentified Caprellidae						1	
Stenothoe valida							
Decapoda							
Crangon nigricauda							
Upogebia pugettensis							**2**
Callianassidae unidentified							
Paguridae (megalopa)	**1**						
Pycnogonida							
Ammotheidae							
Achelia chelata?							
Leptopcarida							
Mollusca							
Gastropoda							
Crepidula convexa	5	29	6				
Haminoea japonica		1					
Philine spp. ***	1	2		1			
Unidentified Nudibranchia			1				

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	STATION						
TAXON	**CB 27**	**CB 28**	**CB 31**	**CB 33**	**CB 35**	**CB 38**	**CB 41**
Unidentified Gastropoda^							
Pelycepoda							
Potamocorbula amurensis							
Cryptomya californica							1
Gemma gemma							
Lyonsia californica							
Macoma balthica/petalum							
Macoma spp.							
Modiolus rectus							
Musculista senhousia		2	4	1			
Mytilus spp.							
Rochefortia tumida							
Siliqua lucida							
Theora lubrica		1	1	3			
Venerupis philippinarum			1				1
Unidentified Bivalvia^							
Bryozoa	+	+					
Chordata							
Molgula spp.							
Metandrocarpa spp. ?							
Total	**134**	**1796**	**598**	**277**	**138**	**235**	**249**

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	STATION						
TAXON	**CB 44**						
Cnidaria							
Hydrozoa							
Anthozoa							
Actiniaria							
Burrowing anenome (#1)	4						
Edwardsiidae (?Scolanthus spp)							
Attached (#2)	1						
Unid. Actiniaria (Diadumene?)							
Alcyonacea - Pennatulidae							
Stylatula spp.							
Turbellaria							
Nemertea							
Tubulanus spp.							
Unidentified Nemertea	3						
Nematoda							
Phoronida							
Phoronis spp.							
Annelida							
Oligochaeta	1						
Naididae							
Tubificidae							
Tubificoides wasselli							
Tubificoides spp.							
Polychaeta							
Tenonia priops							
Harmothoe imbricata	2						
Malmgreniella macginintiei							
Malmgreniella spp. (= Harmothoe spp.)							
Exogone lourei							
Exogone spp.							
Sphaerosyllis californiensis	1						
Sphaerosyllis spp.							
Typosyllis nipponica	2						
Syllidae unidentified	1						
Eteone sp. A							
Phyllodoce longipes	4						
Podarkiopsis glabrus							
Sigambra ?setosa							
Nephtys spp.^^^							
Nephtys caecoides							
Nephtys cornuta							
Glycinde picta (= G. polygnatha)	3						
Glycinde spp.	1						
Dorvillea longicornis (= D. rudolphi)	37						
Dorvillea spp.^^^							

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	STATION						
TAXON	CB 44						
Pettiboneia gracilis							
Lumbrineridae unidentified	3						
Leitoscoloplos pugettensis	3						
Orbiniidae unidentified							
Dipolydora caulleryi							
Polydora cornuta							
Pseudopolydora kempi	1						
Pseudopolydora paucibranchiata	23						
Scolelepis spp.							
Spiophanes bombyx							
Spiophanes spp.							
Streblospio benedicti							
Aricidea spp.							
Chaetozone spp.							
Cirriformia nr. moorei (=sprirabrancha)	11						
Cirratulidae unidentified	1						
Cossura spp.	1						
Acrocirridae unidentified							
Armandia brevis							
Capitella capitata complex							
Heteromastus filiformis	11						
Mediomastus spp.							
Mediomastus ambiseta/californiensis							
Notomastus tenuis							
Capitellidae unid.							
Sabaco elongatus	22						
Maldanidae unidentified							
Euchone limnicola	53						
Ampharete labrops							
Ameana occidentalis							
Ameana sp. SF 1	7						
Ameana spp.****							
Pista spp.							
Polycirrus californicus	1						
Terebellidae unidentified							
Phylum Arthropoda							
Crustacea							
Ostracoda							
Eusarsiella zosteracola							
Copepoda							
Calinoida							
Harpacticoida							
Cirripedia							
Balanidae unidentified							

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	STATION						
TAXON	CB 44						
Mycidacea unidentified							
Cumacea							
Cumella vulgaris							
Eudorella pacifica							
Nippoleucon hinnumensis	8						
Isopoda							
Munna spp.							
Paranthura japonica							
Synidotea laevidorsalis							
Unid. Sphaeromatidae							
Tanaidacea							
Leptochelia spp. (replaces L. dubia)							
Amphipoda							
Ampelisca abdita	95						
Corophium heteroceratum	12						
Corophium spp.							
Grandiderella japonica	1						
Monocorophium acherusicum	1						
Monocorophium insidiosum							
Monocorophium spp.							
Oedicerodidae unidentified							
Photis brevipes*							
Photis spp.**							
Caprella scaura							
Caprella ?verucosa							
Caprella sp. A female							
Caprella spp. (too small to id to species)							
Unidentified Caprellidae							
Stenothoe valida							
Decapoda							
Crangon nigricauda							
Upogebia pugettensis							
Callianassidae unidentified							
Paguridae (megalopa)							
Pycnogonida							
Ammotheidae							
Achelia chelata?							
Leptopcarida							
Mollusca							
Gastropoda							
Crepidula convexa							
Haminoea japonica							
Philine spp. ***							
Unidentified Nudibranchia							

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	STATION						
TAXON	**CB 44**						
Unidentified Gastropoda^	1						
Pelycepoda							
Potamocorbula amurensis							
Cryptomya californica							
Gemma gemma							
Lyonsia californica							
Macoma balthica/petalum							
Macoma spp.							
Modiolus rectus							
Musculista senhousia							
Mytilus spp.							
Rochefortia tumida							
Siliqua lucida							
Theora lubrica							
Venerupis philippinarum							
Unidentified Bivalvia^							
Bryozoa							
Chordata							
Molgula spp.							
Metandrocarpa spp. ?							
Total	**315**						

August 2007 survey

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	STATION						
TAXON	CB 1	CB 2	CB 4	CB05	CB 8	CB 9	CB 15
Cnidaria							
Hydrozoa						+	
Anthozoa							
Actiniaria							
Burrowing anenome (#1)			1				
Burrowing anenome (unid.)				1			
Edwardsiidae (?Scolanthus spp)							
Actiniaria #3							
Diadumene spp.							
Unid. Actiniaria (Diadumene?)							
Attached (#2)			1		15		3
Alcyonacea - Pennatulidae							
Stylatula spp.		2					1
Turbellaria							
Nemertea							
Lineidae unidentified							
Poseidonemertes collaris							
Tubulanus pellucidus							
Tubulanus spp.							
Unidentified Enopla							
Unidentified Nemertea	1	3					2
Nematoda		11			6		3
Phoronida							
Phoronis spp.							
Echiura							
Unidentified Echiura?							
Annelida							
Oligochaeta							
Naididae							
Tubificidae	17						
Tubificoides wasselli							
Tubificoides spp.							
Polychaeta							
Aphrodita spp.							
Tenonia priops		1					
Hesperanoe ?laevis							
Harmothoe imbricata	16	1	3		7	1	6
Malmgreniella macginintiei							
Malmgreniella spp. (= Harmothoe)							
Polynoidea unidentified juveniles							
Pholoe spp.			1				
Sthenelais tetriaglabra							
Exogone lourei		1					
Exogone spp.							
Sphaerosyllis californiensis					1	1	3

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	STATION						
TAXON	CB1	CB 2	CB 4	CB05	CB 8	CB 9	CB 15
Sphaerosyllis spp.	1	3					
Typosyllis nipponica	11	1	1		2	1	5
Autolytinae unidentified							
Syllidae unidentified							
Eteone californica							
Eteone lighti							
Eteone sp. A							
Eteone nr. pigmentata							
Eteone spp.^^							
Eumida sanguinea							
Phyllodoce hartmanae							
Phyllodoce longipes							
Phyllodoce spp.							
Podarkiopsis glabrus							
Sigambra ?setosa							
Unidentified Hesionidae							
Platynereis bicanaliculata							
Neanthes spp. (not N. succinea)							
Nephtys caecoides			1		1		
Nephtys cornuta		3	13				1
Nephtys spp.^^^		1				1	
Nereis latescens							
Neriedae unidentified^^^							
Glycinde picta (= G. polygnatha)	5	3	9	3		4	2
Glycinde spp.	6	2	2		1		
Glycera americana							
Glycera robusta			1				
Onuphis spp.							
Marphysa nr. Sanguinea							
Dorvillea longicornis (= D. rudolphi)	4		4				
Dorvillea spp.^^^							
Pettiboneia gracilis							
Ancistrosyllis groenlandica							
Scoletoma luti							
Scoletoma spp.							
Lumbrineris spp.							
Lumbrineridae unidentified							
Leitoscoloplos pugettensis					2		
Orbiniidae unidentified							
Dipolydora caulleryi (=brachycephala)							
Dipolydora socialis							
Polydora cornuta							
Prionospio lighti							
Pseudopolydora kempi					2		

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	STATION						
TAXON	CB1	CB 2	CB 4	CB5	CB 8	CB 9	CB 15
Pseudopolydora paucibranchiata	11						
Pygospio elegans							
Scolelepis spp.							
Spiophanes berkeleyorum							
Spiophanes bombyx							
Spiophanes duplex							
Spiophanes spp.							
Streblospio benedicti							
Spionidae unidentified^^^							
Aricidea spp.							
Paraonidae unidentified							
Trochochaeta franciscanum							
Aphelochaeta monilaris							
Aphelochaeta spp.							
Chaetozone spp.							
Cirratulus cf. spectabilis							
Cirratulus spp.							
Cirriformia nr. moorei (=sprirabrancha)				3			
Cirriformia spp.							
Tharyx parvus							
Cirratulidae unidentified			2				
Cossura pygodactylata							
Cossura spp.							
Acrocirridae unidentified							
Scalibregma californicum							
Armandia brevis		1	6				
Capitella capitata complex							
Heteromastus filiformis			1			1	
Heteromastus sp. A							
Mediomastus spp.	2						
Mediomastus ambiseta/californiensis							
Notomastus tenuis							
Notomastus sp. A (SCAMIT)							
Capitellidae unid.							2
Sabaco elongatus	3			1	34	1	10
Maldanidae unidentified							
Euchone limnicola	8	14	12		74		27
Sabellidae unidentified						2	
Ampharete acutifrons							
Ampharete labrops							
Ampharete spp.							
Amphicteis scaphobranchiata							
Melinna oculata							
Ameana occidentalis							

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	STATION						
TAXON	**CB1**	**CB 2**	**CB 4**	**CB5**	**CB 8**	**CB 9**	**CB 15**
Ameana sp. SF 1	3						
Ameana spp.****	4	1	1				
Pectinaria californiensis							
Neoamphitrite sp. A							
Pista spp.							
Polycirrus californicus							
Streblosoma sp. SF 1							
Terebellidae unidentified							
Polychaeta unidentified^^^							
Phylum Arthropoda							
Crustacea							
Cephalocarida							
Lightiella serendipita?	1						
Ostracoda							
Cyprideis sp. A							
Eusarsiella zosteracola							
Ostracoda A							
Copepoda							
Calinoida							
Harpacticoida							
Cirripedia							
Balanus spp.							
Balanidae unidentified							
Leptostraca							
Nebaliidae - unidentified							
Mycidacea unidentified							
Cumacea							
Cumella vulgaris							
Eudorella pacifica	7	17	79				
Nippoleucon hinnumensis	36	10	5		45		2
Isopoda							
Munna spp.							
Paranthura japonica							
Synidotea laevidorsalis							
Synidotea spp.							
Unid. Sphaeromatidae							
Tanaidacea							
Leptochelia spp. (replaces L. dubia)			4	14	205	15	33
Sinolobus spp.							
Amphipoda							
Americorophium stimpsoni							
Ampelisca abdita	823	290	129	44	1227	2	131
Americhelidium shoemakeri							
Ampithoe valida							

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	STATION						
TAXON	CB1	CB 2	CB 4	CB5	CB 8	CB 9	CB 15
Ampithoe spp.							
Corophium heteroceratum	79	179	163	18	24	8	4
Corophium spp.							
Grandiderella japonica	20	2	1		7		
Melita spp.							
Melita rylovae							
Monocorophium acherusicum	2					1	
Monocorophium insidiosum							
Monocorophium spp.		1		1	2	1	
Oedicerodidae unidentified		1					
Dyopedos spp.							
Dulichia/Dyopedos spp.							
Gammaridae unid.							
Liljeborjidae unidentified							
Photis brevipes*			2				
Photis spp.**			17	4		3	
Caprella scaura					25	1	18
Caprella ?verucosa							
Caprella sp. A female							
Caprella spp. (too small to id to species)	1						12
Caprella californica							
Caprella ferrea							
Caprella nr. Laeviuscula							
Caprella sp. B male							
Caprella sp. C female							
Tritella pilimana							
Tritella spp.							
Unidentified Caprellidae					9		
Pacificulodes spinnipes	1						
Paradexamine sp.							
Podoceridae^^^							
Stenothoe valida							
Decapoda							
Cancer jordani							1
Crangon nigricauda							
Crangon spp.							
Hemigrapsus oregonensis							
Upogebia pugettensis							
Callianassidae unidentified							
Paguridae (megalopa)							
Pinnotheridae							
Pyromaia tuberculata							
Pycnogonida							
Ammotheidae							

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	STATION						
TAXON	CB1	CB 2	CB 4	CB5	CB 8	CB 9	CB 15
Achelia chelata?							
Insecta							
Diptera							
Chironomidae							
Hemiptera							
Trichocorixa spp.							
Leptopcarida							
Mollusca							
Gastropoda							
Crepidula convexa							
Crepidula plana							
Crepidula spp.							
Haminoea japonica							
Haminoea spp.							
Philine spp. ***	4				1		1
Urosalpinx cinerea							
Unidentified Nudibranchia							
Unidentified Gastropoda^							
Pelycepoda							
Adula digensis?							
Corbula amurensis							
Cryptomya californica						9	
Gemma gemma							
Lyonsia californica				2			
Macoma balthica/petalum							
Macoma spp.							
Modiolus rectus							
Modiolus spp.							
Musculista senhousia				2			2
Mactromeris catilliformis							
Mactridae unidentified^							
Mya arenaria							
Myidae^							
Mytilus spp.	1					1	
Mytilidae unidentified							
Nuculana spp.							
Ostreidae unidentified							
Protothaca spp.							
Rochefortia coani	4						
Rochefortia tumida							
Rochefortia spp.^							
Siliqua lucida							
Siliqua spp.							
Solen spp.							

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	STATION						
TAXON	**CB 1**	**CB 2**	**CB 4**	**CB 5**	**CB 8**	**CB 9**	**CB 15**
Theora lubrica^		1			2		
Venerupis philippinarum							
Veneridae^							
Unidentified Bivalvia^		1		1			1
Bryozoa							
Chordata							
Clavelina spp.							+
Molgula spp.							3
Metandrocarpa spp. ?							
Fish larvae							
Total Abundance	**1071**	**550**	**459**	**94**	**1692**	**53**	**273**
+ Present							

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	STATION						
TAXON	CB 16	CB 17	CB 19	CB 22	CB 24	CB 25	CB 27
Cnidaria							
Hydrozoa							+
Anthozoa							
Actiniaria							
Burrowing anenome (#1)							
Burrowing anenome (unid.)						2	
Edwardsiidae (?Scolanthus spp)		1					
Actiniaria #3							
Diadumene spp.		2					
Unid. Actiniaria (Diadumene?)					45	123	
Attached (#2)	8						122
Alcyonacea - Pennatulidae							
Stylatula spp.	2	4			1		
Turbellaria							
Nemertea							
Lineidae unidentified							
Poseidonemertes collaris							
Tubulanus pellucidus							
Tubulanus spp.							
Unidentified Enopla							
Unidentified Nemertea						1	
Nematoda		1	21		57	64	10
Phoronida					1	35	
Phoronis spp.							
Echiura							
Unidentified Echiura?							
Annelida							
Oligochaeta	13		1				36
Naididae							
Tubificidae		1			13	33	
Tubificoides wasselli							
Tubificoides spp.							
Polychaeta							
Aphrodita spp.							
Tenonia priops							
Hesperanoe ?laevis							
Harmothoe imbricata	9	1	31	1		16	9
Malmgreniella macginintiei							
Malmgreniella spp. (= Harmothoe)							
Polynoidea unidentified juveniles							
Pholoe spp.							
Sthenelais tetriaglabra							
Exogone lourei					182	116	8
Exogone spp.		1					
Sphaerosyllis californiensis	17	3			33	37	

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	STATION						
TAXON	CB 16	CB 17	CB 19	CB 22	CB 24	CB 25	CB 27
Sphaerosyllis spp.			2	1			
Typosyllis nipponica	17		14		2	3	5
Autolytinae unidentified							
Syllidae unidentified						14	
Eteone californica							
Eteone lighti							
Eteone sp. A							
Eteone nr. pigmentata							
Eteone spp.^^							
Eumida sanguinea							
Phyllodoce hartmanae							
Phyllodoce longipes		1					
Phyllodoce spp.							
Podarkiopsis glabrus							
Sigambra ?setosa							
Unidentified Hesionidae							
Platynereis bicanaliculata							
Neanthes spp. (not N. succinea)							
Nephtys caecoides				1			
Nephtys cornuta		2					
Nephtys spp.^^^				3			
Nereis latescens							
Neriedae unidentified^^^							
Glycinde picta (= G. polygnatha)	1			4	3	1	3
Glycinde spp.	4	6	5	3	6	13	5
Glycera americana							
Glycera robusta							
Onuphis spp.							
Marphysa nr. Sanguinea							
Dorvillea longicornis (= D. rudolphi)		17			6	52	54
Dorvillea spp.^^^			3				
Pettiboneia gracilis						1	
Ancistrosyllis groenlandica							
Scoletoma luti							
Scoletoma spp.							
Lumbrineris spp.							
Lumbrineridae unidentified							
Leitoscoloplos pugettensis					1		
Orbiniidae unidentified							
Dipolydora caulleryi (=brachycephala)							
Dipolydora socialis							
Polydora cornuta							
Prionospio lighti							
Pseudopolydora kempi							

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	STATION						
TAXON	CB 16	CB 17	CB 19	CB 22	CB 24	CB 25	CB 27
Pseudopolydora paucibranchiata						23	
Pygospio elegans							
Scolelepis spp.							
Spiophanes berkeleyorum		1					
Spiophanes bombyx							
Spiophanes duplex							
Spiophanes spp.							
Streblospio benedicti							
Spionidae unidentified^^^							
Aricidea spp.							
Paraonidae unidentified							
Trochochaeta franciscanum							
Aphelochaeta monilaris							
Aphelochaeta spp.							
Chaetozone spp.							
Cirratulus cf. spectabilis							
Cirratulus spp.							
Cirriformia nr. moorei (=sprirabrancha)		1					
Cirriformia spp.							
Tharyx parvus							
Cirratulidae unidentified							
Cossura pygodactylata							
Cossura spp.							
Acrocirridae unidentified							
Scalibregma californicum							
Armandia brevis			1				
Capitella capitata complex	2						
Heteromastus filiformis	2		2		2		3
Heteromastus sp. A							
Mediomastus spp.		1					
Mediomastus ambiseta/californiensis							
Notomastus tenuis							
Notomastus sp. A (SCAMIT)							
Capitellidae unid.						1	
Sabaco elongatus	113	3	13				
Maldanidae unidentified							
Euchone limnicola	37	7	29			3	
Sabellidae unidentified							
Ampharete acutifrons							
Ampharete labrops							
Ampharete spp.		1					
Amphicteis scaphobranchiata							
Melinna oculata							
Ameana occidentalis							

DATE: August, 2007							
	STATION						
TAXON	**CB 16**	**CB 17**	**CB 19**	**CB 22**	**CB 24**	**CB 25**	**CB 27**
Ameana sp. SF 1							
Ameana spp.****							
Pectinaria californiensis							
Neoamphitrite sp. A							
Pista spp.							
Polycirrus californicus						8	8
Streblosoma sp. SF 1							
Terebellidae unidentified							
Polychaeta unidentified^^^							
Phylum Arthropoda							
Crustacea							
Cephalocarida							
Lightiella serendipita?							
Ostracoda							
Cyprideis sp. A							
Eusarsiella zosteracola							
Ostracoda A							
Copepoda							
Calinoida							
Harpacticoida							
Cirripedia							
Balanus spp.							
Balanidae unidentified							
Leptostraca							
Nebaliidae - unidentified							
Mycidacea unidentified							
Cumacea							
Cumella vulgaris							
Eudorella pacifica		3		246			
Nippoleucon hinnumensis	91	2	1	8			1
Isopoda							
Munna spp.				1			
Paranthura japonica	3		1		5	14	6
Synidotea laevidorsalis						1	1
Synidotea spp.			1		2		
Unid. Sphaeromatidae							
Tanaidacea							
Leptochelia spp. (replaces L. dubia)	15	7	115	156	13	67	15
Sinolobus spp.							
Amphipoda							
Americorophium stimpsoni							
Ampelisca abdita	1709	43	1112	76		2	5
Americhelidium shoemakeri							
Ampithoe valida							

DATE: August, 2007							
	STATION						
TAXON	CB 16	CB 17	CB 19	CB 22	CB 24	CB 25	CB 27
Ampithoe spp.							
Corophium heteroceratum	47	9	100	37		3	3
Corophium spp.							
Grandiderella japonica	3		7	1	10		
Melita spp.							
Melita rylovae							
Monocorophium acherusicum	18		3		3		1
Monocorophium insidiosum			1			1	
Monocorophium spp.	36				1	20	1
Oedicerodidae unidentified							
Dyopedos spp.							
Dulichia/Dyopedos spp.							
Gammaridae unid.							
Liljeborjidae unidentified							
Photis brevipes*				2			
Photis spp.**		2		42			
Caprella scaura	25		17				
Caprella ?verucosa							
Caprella sp. A female							
Caprella spp. (too small to id to species)	22	1	7			1	
Caprella californica							
Caprella ferrea							
Caprella nr. Laeviuscula							
Caprella sp. B male							
Caprella sp. C female							
Tritella pilimana				1			
Tritella spp.							
Unidentified Caprellidae							
Pacificulodes spinnipes							
Paradexamine sp.							
Podoceridae^^^							
Stenothoe valida							
Decapoda							
Cancer jordani	1						
Crangon nigricauda							
Crangon spp.							
Hemigrapsus oregonensis							
Upogebia pugettensis							
Callianassidae unidentified							
Paguridae (megalopa)							
Pinnotheridae							
Pyromaia tuberculata							
Pycnogonida							
Ammotheidae							

DATE: August, 2007							
	STATION						
TAXON	CB 16	CB 17	CB 19	CB 22	CB 24	CB 25	CB 27
Achelia chelata?							
Insecta							
Diptera							
Chironomidae							
Hemiptera							
Trichocorixa spp.							
Leptopcarida							
Mollusca							
Gastropoda							
Crepidula convexa						175	43
Crepidula plana						1	
Crepidula spp.							
Haminoea japonica							
Haminoea spp.							
Philine spp. ***				1		3	
Urosalpinx cinerea							
Unidentified Nudibranchia							
Unidentified Gastropoda^							
Pelycepoda							
Adula digensis?							
Corbula amurensis							1
Cryptomya californica					1		
Gemma gemma					1		
Lyonsia californica		6			2		
Macoma balthica/petalum							
Macoma spp.							
Modiolus rectus				4			
Modiolus spp.							
Musculista senhousia	3		1		3	7	3
Mactromeris catilliformis							
Mactridae unidentified^		1					
Mya arenaria				1			
Myidae^							
Mytilus spp.		1					
Mytilidae unidentified							
Nuculana spp.						1	
Ostreidae unidentified							
Protothaca spp.							
Rochefortia coani							
Rochefortia tumida							
Rochefortia spp.^							
Siliqua lucida							
Siliqua spp.							
Solen spp.							

DATE: August, 2007							
	STATION						
TAXON	**CB 16**	**CB 17**	**CB 19**	**CB 22**	**CB 24**	**CB 25**	**CB 27**
Theora lubrica^							
Venerupis philippinarum					17		2
Veneridae^							
Unidentified Bivalvia^		4			1		
Bryozoa						+	+
Chordata							
Clavelina spp.							
Molgula spp.	3					20	13
Metandrocarpa spp. ?							
Fish larvae							
Total Abundance	**2201**	**133**	**1488**	**589**	**411**	**862**	**358**
+ Present							

DATE: August, 2007							
	STATION						
TAXON	CB 28	CB 33	CB 35	CB 39	CB 40	CB 41	CB 44
Cnidaria							
Hydrozoa							
Anthozoa							
Actiniaria							
Burrowing anenome (#1)							
Burrowing anenome (unid.)							
Edwardsiidae (?Scolanthus spp)							
Actiniaria #3							
Diadumene spp.							
Unid. Actiniaria (Diadumene?)							
Attached (#2)	72				1		
Alcyonacea - Pennatulidae							
Stylatula spp.			1	1	5		1
Turbellaria							
Nemertea							
Lineidae unidentified							
Poseidonemertes collaris							
Tubulanus pellucidus							
Tubulanus spp.							
Unidentified Enopla							
Unidentified Nemertea						1	
Nematoda	23		30	55	16	166	
Phoronida							
Phoronis spp.							
Echiura							
Unidentified Echiura?							
Annelida							
Oligochaeta	18				1		
Naididae							
Tubificidae							
Tubificoides wasselli							
Tubificoides spp.							
Polychaeta							
Aphrodita spp.							
Tenonia priops							
Hesperanoe ?laevis							
Harmothoe imbricata	19	14	7	2	5	8	1
Malmgreniella macginintiei							
Malmgreniella spp. (= Harmothoe)							
Polynoidea unidentified juveniles							
Pholoe spp.							
Sthenelais tetriaglabra							
Exogone lourei	95				1		
Exogone spp.							
Sphaerosyllis californiensis	17	3			33	37	

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	STATION						
TAXON	CB 28	CB 33	CB 35	CB 39	CB 40	CB 41	CB 44
Sphaerosyllis spp.							
Typosyllis nipponica	7	2	4		5	8	2
Autolytinae unidentified							
Syllidae unidentified	3						
Eteone californica							
Eteone lighti							
Eteone sp. A							
Eteone nr. pigmentata							
Eteone spp.^^							
Eumida sanguinea							
Phyllodoce hartmanae							
Phyllodoce longipes							
Phyllodoce spp.							
Podarkiopsis glabrus							
Sigambra ?setosa							
Unidentified Hesionidae							
Platynereis bicanaliculata							
Neanthes spp. (not N. succinea)							
Nephtys caecoides							
Nephtys cornuta							
Nephtys spp.^^^							
Nereis latescens							
Neriedae unidentified^^^							
Glycinde picta (= G. polygnatha)	6	1					
Glycinde spp.	2						
Glycera americana							
Glycera robusta							
Onuphis spp.		1					
Marphysa nr. Sanguinea							
Dorvillea longicornis (= D. rudolphi)							
Dorvillea spp.^^^	32	1	17		13	3	14
Pettiboneia gracilis							
Ancistrosyllis groenlandica							
Scoletoma luti							
Scoletoma spp.							
Lumbrineris spp.							
Lumbrineridae unidentified					1		2
Leitoscoloplos pugettensis		4	3		10	5	
Orbiniidae unidentified							
Dipolydora caulleryi (=brachycephala)					1		
Dipolydora socialis		1					
Polydora cornuta							
Prionospio lighti							
Pseudopolydora kempi						1	

DATE: August, 2007							
	STATION						
TAXON	CB 28	CB 33	CB 35	CB 39	CB 40	CB 41	CB 44
Pseudopolydora paucibranchiata						23	
Pygospio elegans							
Scolelepis spp.							
Spiophanes berkeleyorum		1					
Spiophanes bombyx							
Spiophanes duplex							
Spiophanes spp.							
Streblospio benedicti							
Spionidae unidentified^^^							
Aricidea spp.							
Paraonidae unidentified							
Trochochaeta franciscanum							
Aphelochaeta monilaris							
Aphelochaeta spp.							
Chaetozone spp.							
Cirratulus cf. spectabilis							
Cirratulus spp.							
Cirriformia nr. moorei (=sprirabrancha)		1					
Cirriformia spp.							
Tharyx parvus							
Cirratulidae unidentified							
Cossura pygodactylata							
Cossura spp.							
Acrocirridae unidentified							
Scalibregma californicum							
Armandia brevis			1				
Capitella capitata complex	2						
Heteromastus filiformis	2		2		2		3
Heteromastus sp. A							
Mediomastus spp.		1					
Mediomastus ambiseta/californiensis							
Notomastus tenuis							
Notomastus sp. A (SCAMIT)							
Capitellidae unid.						1	
Sabaco elongatus	113	3	13				
Maldanidae unidentified							
Euchone limnicola	37	7	29			3	
Sabellidae unidentified							
Ampharete acutifrons							
Ampharete labrops							
Ampharete spp.		1					
Amphicteis scaphobranchiata							
Melinna oculata							
Ameana occidentalis	6		232			26	6

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TAXON							
Ameana sp. SF 1							
Ameana spp.****							
Pectinaria californiensis							
Neoamphitrite sp. A							
Pista spp.	2					1	
Polycirrus californicus							
Streblosoma sp. SF 1			1				
Terebellidae unidentified							
Polychaeta unidentified^^^							
Phylum Arthropoda							
Crustacea							
Cephalocarida							
Lightiella serendipita?							
Ostracoda							
Cyprideis sp. A							
Eusarsiella zosteracola							
Ostracoda A							
Copepoda							
Calinoida							
Harpacticoida							
Cirripedia							
Balanus spp.							
Balanidae unidentified							
Leptostraca							
Nebaliidae - unidentified							
Mycidacea unidentified							
Cumacea							
Cumella vulgaris							
Eudorella pacifica		50	7	24	2	9	
Nippoleucon hinnumensis							
Isopoda							
Munna spp.	7	2	1			1	1
Paranthura japonica							
Synidotea laevidorsalis							
Synidotea spp.							
Unid. Sphaeromatidae							
Tanaidacea					3		
Leptochelia spp. (replaces L. dubia)							
Sinolobus spp.							
Amphipoda							
Americorophium stimpsoni	4	379	8	3	21	13	10
Ampelisca abdita							
Americhelidium shoemakeri							
Ampithoe valida							

DATE: August, 2007							
	STATION						
TAXON	CB 28	CB 33	CB 35	CB 39	CB 40	CB 41	CB 44
Ampithoe spp.		25	3		3	15	
Corophium heteroceratum							
Corophium spp.		2	1	1	2	2	1
Grandiderella japonica							
Melita spp.							
Melita rylovae		1	2				
Monocorophium acherusicum							
Monocorophium insidiosum	1	7					
Monocorophium spp.							
Oedicerodidae unidentified							
Dyopedos spp.							
Dulichia/Dyopedos spp.							
Gammaridae unid.							
Liljeborjidae unidentified							
Photis brevipes*					2		
Photis spp.**		7	1		1		1
Caprella scaura							
Caprella ?verucosa							
Caprella sp. A female							
Caprella spp. (too small to id to species)							
Caprella californica							
Caprella ferrea							
Caprella nr. Laeviuscula							
Caprella sp. B male							
Caprella sp. C female							
Tritella pilimana							
Tritella spp.				2			
Unidentified Caprellidae							
Pacificulodes spinnipes							
Paradexamine sp.							
Podoceridae^^^							
Stenothoe valida							
Decapoda							
Cancer jordani							
Crangon nigricauda							
Crangon spp.							
Hemigrapsus oregonensis							
Upogebia pugettensis							
Callianassidae unidentified							
Paguridae (megalopa)		1					
Pinnotheridae							
Pyromaia tuberculata							
Pycnogonida							
Ammotheidae		25	3		3	15	

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	STATION						
TAXON	CB 28	CB 33	CB 35	CB 39	CB 40	CB 41	CB 44
Achelia chelata?							
Insecta							
Diptera							
Chironomidae							
Hemiptera							
Trichocorixa spp.							
Leptopcarida							
Mollusca							
Gastropoda							
Crepidula convexa	58						
Crepidula plana	6						
Crepidula spp.	4						
Haminoea japonica							
Haminoea spp.							
Philine spp. ***							1
Urosalpinx cinerea							
Unidentified Nudibranchia							
Unidentified Gastropoda^							
Pelycepoda							
Adula digensis?							
Corbula amurensis							
Cryptomya californica							
Gemma gemma				2			
Lyonsia californica			1	1			
Macoma balthica/petalum							
Macoma spp.							
Modiolus rectus							
Modiolus spp.							
Musculista senhousia		3	2	1	1		1
Mactromeris catilliformis							
Mactridae unidentified^							
Mya arenaria							
Myidae^							
Mytilus spp.							
Mytilidae unidentified							
Nuculana spp.							
Ostreidae unidentified							
Protothaca spp.							
Rochefortia coani							
Rochefortia tumida							
Rochefortia spp.^							
Siliqua lucida							
Siliqua spp.							
Solen spp.							

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	STATION						
TAXON	CB 28	CB 33	CB 35	CB 39	CB 40	CB 41	CB 44
Theora lubrica^		3			6	3	3
Venerupis philippinarum							
Veneridae^							
Unidentified Bivalvia^					1		
Bryozoa							
Chordata							
Clavelina spp.							
Molgula spp.	6		17				
Metandrocarpa spp. ?							
Fish larvae							
Total Abundance	**387**	**560**	**444**	**94**	**144**	**299**	**180**
+ Present							

DATE: August, 2007							
	STATION						
TAXON	CB 48	CB 49	CB 50	CB 51	CB 52	CB 53	
Cnidaria							
Hydrozoa							
Anthozoa							
Actiniaria							
Burrowing anenome (#1)						2	
Burrowing anenome (unid.)							
Edwardsiidae (?Scolanthus spp)							
Actiniaria #3							
Diadumene spp.							
Unid. Actiniaria (Diadumene?)							
Attached (#2)			2	2			
Alcyonacea - Pennatulidae							
Stylatula spp.							
Turbellaria							
Nemertea			2				
Lineidae unidentified							
Poseidonemertes collaris							
Tubulanus pellucidus							
Tubulanus spp.							
Unidentified Enopla							
Unidentified Nemertea						2	
Nematoda	794	467	141	11	185	798	
Phoronida							
Phoronis spp.							
Echiura							
Unidentified Echiura?							
Annelida							
Oligochaeta	43	5		214	1	119	
Naididae							
Tubificidae							
Tubificoides wasselli							
Tubificoides spp.							
Polychaeta							
Aphrodita spp.							
Tenonia priops							
Hesperanoe ?laevis							
Harmothoe imbricata			6	3			
Malmgreniella macginintiei							
Malmgreniella spp. (= Harmothoe)							
Polynoidea unidentified juveniles							
Pholoe spp.							
Sthenelais tetriaglabra							
Exogone lourei	7	17		15		1	
Exogone spp.							
Sphaerosyllis californiensis	17	3			33	37	

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	STATION						
TAXON	CB 48	CB 49	CB 50	CB 51	CB 52	CB 53	
Sphaerosyllis spp.							
Typosyllis nipponica			4	2			
Autolytinae unidentified							
Syllidae unidentified				1			
Eteone californica							
Eteone lighti						6	
Eteone sp. A						13	
Eteone nr. pigmentata							
Eteone spp.^^		1					
Eumida sanguinea							
Phyllodoce hartmanae							
Phyllodoce longipes							
Phyllodoce spp.							
Podarkiopsis glabrus							
Sigambra ?setosa							
Unidentified Hesionidae							
Platynereis bicanaliculata							
Neanthes spp. (not N. succinea)	1						
Nephtys caecoides							
Nephtys cornuta							
Nephtys spp.^^^							
Nereis latescens							
Neriedae unidentified^^^							
Glycinde picta (= G. polygnatha)	12	6	15	10	7	8	
Glycinde spp.	15	8	13	33	2		
Glycera americana							
Glycera robusta							
Onuphis spp.							
Marphysa nr. Sanguinea							
Dorvillea longicornis (= D. rudolphi)							
Dorvillea spp.^^^							
Pettiboneia gracilis							
Ancistrosyllis groenlandica							
Scoletoma luti							
Scoletoma spp.							
Lumbrineris spp.							
Lumbrineridae unidentified							
Leitoscoloplos pugettensis			1				
Orbiniidae unidentified							
Dipolydora caulleryi (=brachycephala)							
Dipolydora socialis							
Polydora cornuta					4		
Prionospio lighti							
Pseudopolydora kempi				3	4		

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	STATION						
TAXON	CB 48	CB 49	CB 50	CB 51	CB 52	CB 53	
Pseudopolydora paucibranchiata				10		1	
Pygospio elegans							
Scolelepis spp.							
Spiophanes berkeleyorum							
Spiophanes bombyx							
Spiophanes duplex							
Spiophanes spp.							
Streblospio benedicti				4	2		
Spionidae unidentified^^^			2		2		
Aricidea spp.							
Paraonidae unidentified							
Trochochaeta franciscanum							
Aphelochaeta monilaris							
Aphelochaeta spp.							
Chaetozone spp.							
Cirratulus cf. spectabilis							
Cirratulus spp.							
Cirriformia nr. moorei (=sprirabrancha)			1		4	4	
Cirriformia spp.							
Tharyx parvus							
Cirratulidae unidentified							
Cossura pygodactylata							
Cossura spp.							
Acrocirridae unidentified							
Scalibregma californicum							
Armandia brevis							
Capitella capitata complex			1		1	3	
Heteromastus filiformis							
Heteromastus sp. A							
Mediomastus spp.							
Mediomastus ambiseta/californiensis							
Notomastus tenuis							
Notomastus sp. A (SCAMIT)							
Capitellidae unid.							
Sabaco elongatus							
Maldanidae unidentified							
Euchone limnicola				2	1		
Sabellidae unidentified							
Ampharete acutifrons							
Ampharete labrops							
Ampharete spp.							
Amphicteis scaphobranchiata							
Melinna oculata							
Ameana occidentalis							

DATE: August, 2007							
	STATION						
TAXON	CB 48	CB 49	CB 50	CB 51	CB 52	CB 53	
Ameana sp. SF 1							
Ameana spp.****							
Pectinaria californiensis							
Neoamphitrite sp. A							
Pista spp.							
Polycirrus californicus			1				
Streblosoma sp. SF 1							
Terebellidae unidentified							
Polychaeta unidentified^^^							
Phylum Arthropoda							
Crustacea							
Cephalocarida							
Lightiella serendipita?							
Ostracoda							
Cyprideis sp. A						8	
Eusarsiella zosteracola	1				2	19	
Ostracoda A		1					
Copepoda							
Calinoida							
Harpacticoida							
Cirripedia							
Balanus spp.	3						
Balanidae unidentified							
Leptostraca							
Nebaliidae - unidentified			3				
Mycidacea unidentified							
Cumacea							
Cumella vulgaris							
Eudorella pacifica							
Nippoleucon hinnumensis	21	3	6	10			
Isopoda							
Munna spp.							
Paranthura japonica		2	15	4	1	4	
Synidotea laevidorsalis							
Synidotea spp.							
Unid. Sphaeromatidae							
Tanaidacea							
Leptochelia spp. (replaces L. dubia)							
Sinolobus spp.							
Amphipoda							
Americorophium stimpsoni				189			
Ampelisca abdita	2		5	38	1		
Americhelidium shoemakeri							
Ampithoe valida			1				

DATE: August, 2007							
	STATION						
TAXON	CB 48	CB 49	CB 50	CB 51	CB 52	CB 53	
Ampithoe spp.			1				
Corophium heteroceratum	1						
Corophium spp.							
Grandiderella japonica	43	68	119	71	58	311	
Melita spp.			1				
Melita rylovae							
Monocorophium acherusicum	1	2	6	27			
Monocorophium insidiosum	2		13	1	2	6	
Monocorophium spp.	3	3	59	26			
Oedicerodidae unidentified		1	1				
Dyopedos spp.							
Dulichia/Dyopedos spp.							
Gammaridae unid.							
Liljeborjidae unidentified							
Photis brevipes*							
Photis spp.**							
Caprella scaura							
Caprella ?verucosa							
Caprella sp. A female							
Caprella spp. (too small to id to species)					2		
Caprella californica							
Caprella ferrea							
Caprella nr. Laeviuscula							
Caprella sp. B male							
Caprella sp. C female							
Tritella pilimana							
Tritella spp.							
Unidentified Caprellidae						2	
Pacificulodes spinnipes							
Paradexamine sp.							
Podoceridae^^^							
Stenothoe valida							
Decapoda							
Cancer jordani							
Crangon nigricauda							
Crangon spp.					1		
Hemigrapsus oregonensis							
Upogebia pugettensis							
Callianassidae unidentified							
Paguridae (megalopa)							
Pinnotheridae							
Pyromaia tuberculata							
Pycnogonida							
Ammotheidae							

DATE: August, 2007							
	STATION						
TAXON	CB 48	CB 49	CB 50	CB 51	CB 52	CB 53	
Achelia chelata?							
Insecta							
Diptera							
Chironomidae							
Hemiptera							
Trichocorixa spp.							
Leptopcarida							
Mollusca							
Gastropoda							
Crepidula convexa							
Crepidula plana							
Crepidula spp.							
Haminoea japonica			7		1		
Haminoea spp.						1	
Philine spp. ***	1		1	2			
Urosalpinx cinerea							
Unidentified Nudibranchia							
Unidentified Gastropoda^							
Pelycepoda							
Adula digensis?							
Corbula amurensis	1						
Cryptomya californica							
Gemma gemma	1				5005	12082	
Lyonsia californica							
Macoma balthica/petalum	1						
Macoma spp.							
Modiolus rectus							
Modiolus spp.							
Musculista senhousia	1	2				7	
Mactromeris catilliformis							
Mactridae unidentified^							
Mya arenaria							
Myidae^							
Mytilus spp.							
Mytilidae unidentified							
Nuculana spp.							
Ostreidae unidentified							
Protothaca spp.							
Rochefortia coani							
Rochefortia tumida							
Rochefortia spp.^							
Siliqua lucida							
Siliqua spp.							
Solen spp.							

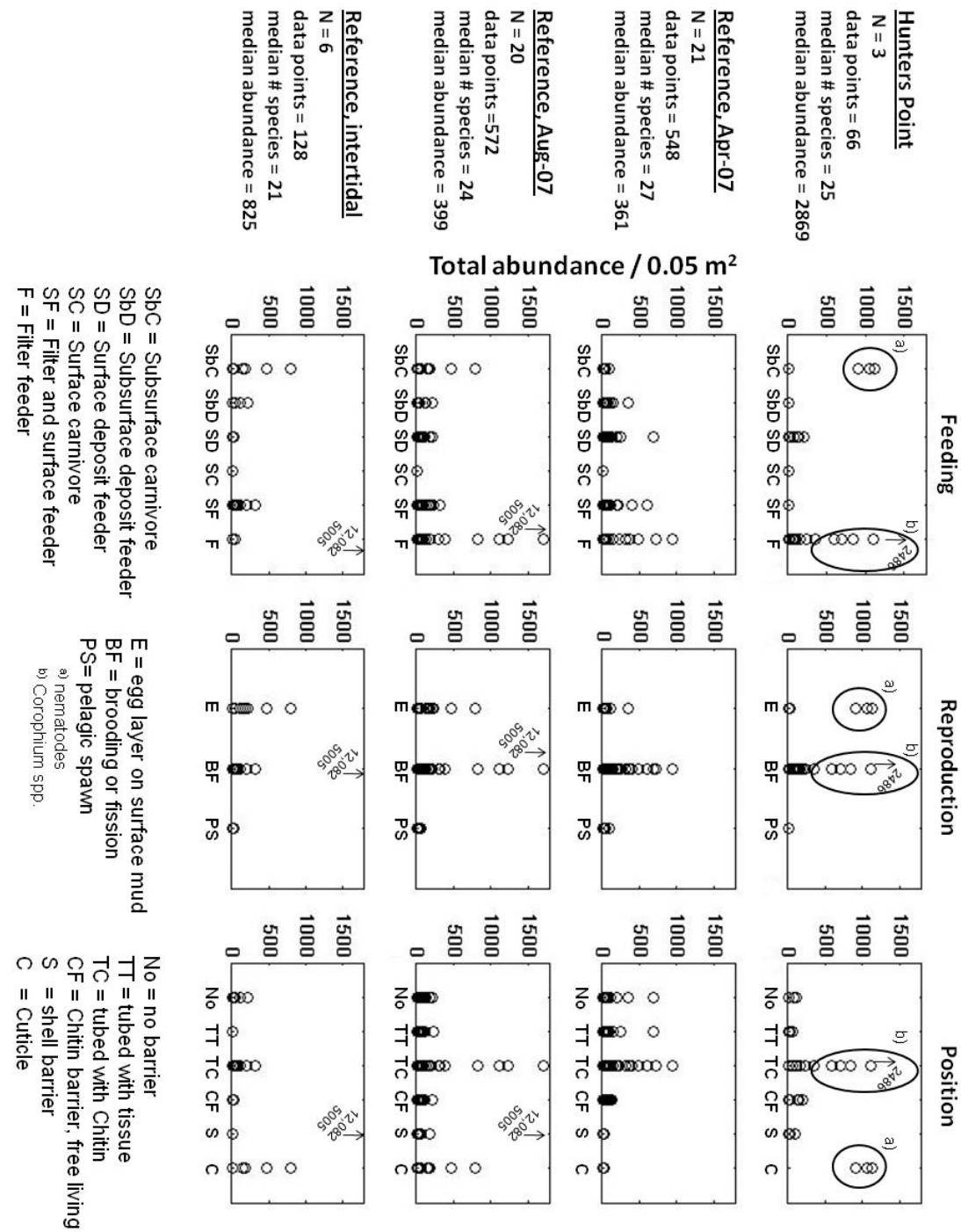
DATE: August, 2007							
	STATION						
TAXON	CB 48	CB 49	CB 50	CB 51	CB 52	CB 53	
Theora lubrica^							
Venerupis philippinarum	4	7		1	3	7	
Veneridae^							
Unidentified Bivalvia^							
Bryozoa							
Chordata							
Clavelina spp.							
Molgula spp.							
Metandrocarpa spp. ?							
Fish larvae							
Total Abundance	**958**	**593**	**427**	**692**	**5289**	**13404**	
+ Present							

DATE: September 2008							
	STATION - Hunters Point						
TAXON	**1**	**2**	**3**				
Cnidaria							
Anthozoa							
Actiniaria - attached			10				
Actiniaria - burrowing anenome	15						
Nematoda	1058	906	1123				
Nemertea	2						
Polychaeta							
Exogone lourei	73		115				
Typosyllis nipponica	1	2	10				
Glycinde picta	1						
Dipolydora socialis		5	19				
Pseudopolydora paucibranchiata	8	5	10				
Polydorinae unidentified	6						
Cirriformia nr. Moorei		1					
Cirratulidae unidentified	1						
Capitella capitata complex		2					
Heteromastus filiformis		1					
Phylum Arthropoda							
Crustacea							
Ostracoda							
Eusarsiella zosteracola	8	8	19				
Ostracoda unidentified	1						
Cumacea							
Nippoleucon hinnumensis	3	5					
Isopoda							
Sphaeromatidae - unidentified		1					
Paranthura japonica	143	199	125				
Amphipoda							
Ampithoe spp.	1	1					
Grandidierella japonica	347	234	163				
Monocorophium acherusicum	16	15	845				
Monocorophium insidiosum	64	113	2486				
Monocorophium spp.	1105	593	701				
Caprella scaura		1					
Caprellidae - unidentified		1					
Insecta							
Chironomidae							
Orthocladiinae	2	1	10				
Chironomidae unidentified pupa	1	1					

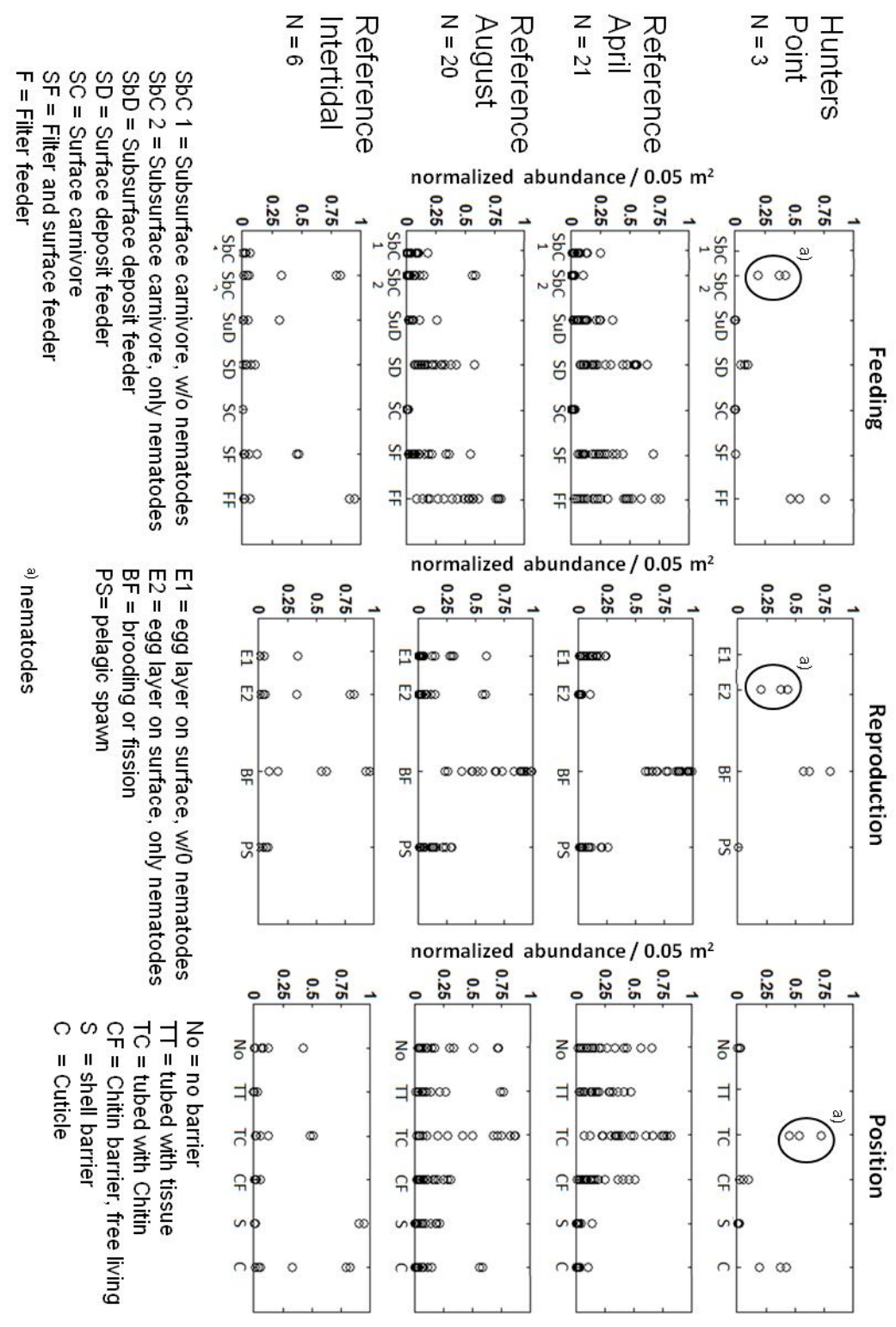
DATE: September 2008							
	STATION - Hunters Point						
TAXON	**1**	**2**	**3**				
Mollusca							
Gastropoda							
Haminoea ?japonica		1					
Pelycepoda							
Gemma gemma	11	19	86				
Lyonsia californica		1					
Musculista senhousia	1	1	10				
Venerupis philippinarum	1	8	10				
Total	2869	2125	5741				

5. Total and normalized abundance plots

Total abundance



Normalized Abundance



6. Data for Figure 31

PCB tissue concentrations of *N. arenaceodentata* for contaminants accumulated from untreated Hunters Point Sediment. (Janssen et al. 2010)

Homolog group	**Tetra**		**Penta**		**Hexa**		**Hepta**		**Octa**	
[ng/g]										
exposure time	average	std dev	average	std dev	average	std dev	average	std dev	average	std dev
day 0	313.7	343.0	158.1	55.3	295.1	186.2	22.6	8.2	4.5	6.2
day 7	113.5	63.8	143.8	33.8	392.5	114.3	121.9	75.9	44.7	53.0
day 14	107.9	77.4	283.3	151.0	696.9	167.2	548.0	398.0	146.5	188.7
day 21	245.2	48.5	760.3	157.2	1571.4	414.0	1599.6	399.4	340.5	74.3
day 28	355.1	82.3	1067.4	274.5	2232.7	514.0	2336.2	581.0	527.2	132.4

PCB tissue concentrations of *N. arenaceodentata* for contaminants accumulated from AC-amended Hunters Point Sediment. (Janssen et al. 2010)

Homolog group	**Tetra**		**Hexa**		**Penta**		**Hepta**		**Octa**	
[ng/g]										
depuration time	average	std dev	average	std dev	average	std dev	average	std dev	average	std dev
day 0	276.7	221.4	105.0	61.5	51.4	20.7	10.7	6.1	19.0	11.3
day 7	237.7	173.9	53.5	38.9	96.2	78.7	101.8	73.7	50.2	46.4
day 14	115.0	55.7	40.9	29.6	103.9	68.7	139.5	83.6	81.0	32.2
day 21	83.3	49.8	35.3	19.9	64.1	36.0	108.3	68.7	74.2	55.7
day 28	76.6	11.2	28.6	2.8	62.2	19.6	88.9	29.1	60.0	23.6

7. Data for Figure 34

PCB tissue concentrations of *N. arenaceodentata* for contaminants accumulated from untreated Hunters Point Sediment. (Janssen et al. 2011)

Test ID	**Ex-situ**		**In-situ (Feb)**		**In-situ (July)**	
[ng/g]	Average	std dev	average	std dev	average	std dev
Homolog group						
Tetra	79.0	3.1	215.2	27.1	47.6	9.6
Penta	646.7	77.0	377.6	42.6	372.1	85.8
Hexa	1006.2	149.3	573.5	55.2	581.5	159.3
Hepta	1090.2	158.4	483.3	99.8	531.6	127.3
Octa	334.5	48.2	172.7	25.9	174.3	37.4

PCB tissue concentrations of *N. arenaceodentata* for contaminants accumulated from AC-amended Hunters Point Sediment. (Janssen et al. 2011)

Test ID	**Ex-situ**		**In-situ (Feb)**		**In-situ (July)**	
[ng/g]	average	std dev	average	std dev	average	std dev
Homolog group						
Tetra	18.9	4.5	111.2	23.3	38.8	24.5
Penta	53.5	3.6	202.5	30.4	267.4	98.2
Hexa	82.7	16.4	295.5	44.8	409.6	169.5
Hepta	107.7	17.3	270.4	48.1	358.8	131.3
Octa	53.4	7.4	95.8	35.2	121.2	31.9

8. Data for Figure 35

PCB concentrations in polyoxymethylene samplers after 14-day deployment in untreated and AC-amended Hunters Point Sediment. (Janssen et al. 2011)

[mg/kg]	**AC-amendment**		**untreated sediment**	
deployment	average	std dev	average	std dev
Subsurface	0.45	0.06	0.07	0.01
Subsurface	0.63	0.13	0.13	0.03
Surface	0.51	0.04	0.32	0.1

9. Data for Figure 28

Stations ID	no barrier and tubed with tissue	egg layer w/o nematodes	deposit feeder	subsurface carnivores w/o nematodes
reference sites [April]				
CB1	12.8	0.6	26.6	0.1
CB2	5.7	0.0	23.3	1.3
CB4	13.0	0.0	18.6	1.1
CB5	19.7	0.0	58.8	1.8
CB8	15.3	3.0	22.6	2.3
CB9	18.1	1.4	48.4	1.8
CB12	31.3	0.0	32.0	8.8
CB15	10.8	0.9	13.7	0.0
CB16	14.7	0.0	19.0	1.5
CB17	42.2	1.1	75.1	0.0
CB20	36.8	0.0	42.9	4.3
CB24	62.0	0.0	58.9	1.9
CB25	58.4	10.1	56.3	0.0
CB27	59.0	1.5	35.8	15.3
CB28	84.4	1.8	76.7	24.3
CB31	40.5	0.8	41.0	6.5
CB33	40.4	0.0	56.3	7.5
CB35	55.8	0.0	47.1	2.9
CB38	54.9	0.0	32.8	4.3
CB41	39.4	2.0	45.0	10.3
CB44	62.5	0.0	23.8	3.2
references sites [August]				
CB1	9.0	0.0	9.5	1.4
CB2	6.7	2.0	7.6	0.9
CB4	12.9	0.0	29.2	3.5
CB5	8.5	0.0	23.4	3.2
CB8	8.3	0.4	19.7	0.1
CB9	22.6	0.0	47.2	7.5
CB15	24.2	1.1	33.7	0.7
CB16	10.4	0.0	14.9	0.2
CB17	39.8	0.8	22.6	17.3
CB19	6.8	1.4	13.8	0.5
CB24	71.8	13.9	61.6	3.6
CB25	58.5	7.4	36.2	7.7
CB27	74.3	2.8	25.7	17.3
CB28	73.4	5.9	43.2	10.3
CB33	14.2	0.0	18.1	0.5
CB35	87.4	6.8	16.0	3.8
CB39	5.3	58.5	31.9	0.0
CB40	59.7	11.1	32.6	9.0
CB41	30.1	55.5	14.7	1.0
CB44	90.6	0.0	31.7	7.8
Intertidal Stations				
CB48	8.2	82.9	7.6	2.8
CB49	6.2	78.8	4.9	2.4
CB50	13.1	33.0	11.2	6.6
CB51	45.4	1.6	38.4	6.2
CB52	0.5	3.5	0.3	0.2
CB53	1.2	6.0	1.2	0.1
Hunters Point				
HP2	3.7	0.6	8.2	0.0
HP3	2.0	0.6	10.7	0.0
HP5	4.0	0.5	5.1	0.0